Testing and developing tools for macrophyte management in small Canterbury agricultural waterways

Thesis submitted for the degree of

Doctor of Philosophy in Biological Sciences

Kathryn Elizabeth Collins

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School of Biological Sciences

University of Canterbury

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“Weeds are flowers too,
once you get to know them”
– A.A. Milne

Contents

Abstract

Aquatic macrophytes provide important functions in fresh waters. However, excessive growth in small lowland agricultural streams, particularly of introduced macrophytes, can have negative impacts. These include accumulating sediment and causing flooding of farmland. Many New Zealand farmers see drainage as the primary function of small agricultural streams, and they are often referred to as “drains”. These drains are perceived to be primarily for removing floodwaters and high flows as efficiently as possible and are considered to have little ecological value, despite studies showing the contrary.

During summer, agricultural drains can become choked with macrophytes requiring local water management agencies to control their growth. Conventional macrophyte control techniques, including mechanical clearance and chemical sprays, have a range of adverse effects, such as over-steepening of banks, damaging to in-stream habitat, causing death of native fishes, spreading weed fragments and hindering aquatic ecosystem function. My thesis investigated some of the factors that influence macrophyte diversity, abundance and biomass in agricultural streams in Canterbury and evaluated the effectiveness of alternative practical macrophyte control options for small farm waterways.

To investigate the factors that influence macrophyte species diversity and growth (percent cover), I undertook a region-wide field survey of 28 small streams (<5 m wetted width) across the Canterbury region, South Island, New Zealand. Sites were surveyed at both the stream (i.e. 50 m) and patch scale (i.e. 1 m). Overall, macrophyte diversity was very low with only thirteen species in total found. Streams were dominated by the introduced *Erythranthe guttata* (monkey musk) and *Nasturtium microphyllum* (watercress). Physical and chemical conditions and macrophyte cover varied greatly both between streams and at the patch scale. At the stream scale, I found a significant positive relationship between macrophyte and sediment cover and significant negative relationships between macrophyte cover and both water temperature and dissolved oxygen saturation. At the patch scale, significant positive relationships were recorded between macrophyte and sediment cover, sediment depth and distance to nearest plants and

significant negative relationships between macrophyte cover and stream shade and water velocity. I present a conceptual model of factors influencing macrophyte distribution and growth operating at the stream and patch scale. At the stream scale, my work indicates that disturbance regime is the key factor limiting macrophyte growth, compared to shading at the patch scale. Improving our understanding of these factors which influence macrophyte abundance and success is helpful in terms of informing alternative management regimes to manage excessive growth in lowland streams.

Given the known adverse effects of conventional macrophyte control techniques, I undertook several small- (2 m and < 5 m), reach- (50 m) and large-scale trials (up to 400 m), to evaluate the effectiveness of alternative macrophyte control tools. Alternative control techniques tested included: hand weeding; herbicide spray; weed mat; flower and seed removal; shading; physical disturbance; and sediment removal. At a small-scale, hand weeding, weed matting and herbicide spraying were effective at reducing macrophyte cover to <5 %. Hand weeding and weed mat immediately reduced cover, while dieback from herbicide took two months. Weed mat was a novel and effective control mechanism, particularly for sprawling emergent macrophytes which are rooted in stream banks. In contrast, macrophyte growth was enhanced under a partially shaded channel; whereas, in a subsequent more intensive trial with full shading (80 % light reduction), cover was reduced from almost 100 % to 17 % within six months. In the reach- to large-scale trials, both artificial shading and weed mat also proved to be very effective macrophyte control techniques. Furthermore, although large-scale intensive hand weeding provided short-term control, it proved not to be an effective long-term control option. The combination of weed mat and shading provides effective short- and long-term macrophyte control. Weed mat is practical and effective to suppress macrophyte growth while new riparian plantings grow and establish to provide the necessary shade that ensures continued macrophyte control. There is clearly some value in considering alternative tools to effectively control macrophytes in agricultural streams.

Identifying “drains” as “streams” and recognising that they provide important contributions to overall fresh water ecosystem health, further promotes the case for widespread implementation of alternative control methods.

Plate 1. A spring-fed lowland agricultural headwater stream impacted by excessive macrophyte growth, high levels of dissolved nutrients, fine sediment and faecal bacteria, and low biodiversity.

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Chapter 1:
General introduction to macrophyte management in small agricultural streams

Global setting

Globally, human activities including deforestation, irrigation, land-use conversion and intensification have altered ecosystem function and biodiversity in freshwater systems (Dudgeon et al. 2006; Vörösmarty et al. 2010). These actions have degraded environmental conditions, altered spatial connections in rivers and enabled the spread of pest plants and animals. Fresh waters are now among the most threatened landscapes in the world (Reid et al. 2005; Dudgeon et al. 2006). Among the challenges facing water managers internationally is the management of freshwater introduced plants which have become ubiquitous, particularly in lowland, agricultural waterways.

New Zealand context

Widespread clearance of native bush has been undertaken since human settlement in New Zealand, such that indigenous forest now covers only 24 % of the total land area (Ewers et al. 2006). In comparison, 40 % has been converted to exotic pasture grass species grazed by ruminant animals (including cattle, sheep and deer) (Scarsbrook et al. 2016). Farming is now the most common land use in the middle to lower catchments of many New Zealand rivers (Storey and Cowley 1997; Quinn 2000).

Sheep numbers peaked at 70 million in the early 1980s; however, since then significant intensification of dairy farming has occurred. This has resulted in declining numbers of sheep (from 50 million in 1994 to 30.8 million in 2013) and beef cattle (from 5 million in 1994 to 3.7 million in 2012), but increasing numbers of dairy cattle (from 3.8 million in 1994 to 6.5 million in 2012) (Scarsbrook et al. 2016). The substantial increase in dairy farming has been especially pronounced in the South Island, with stock numbers increasing from 0.5 million in 1994 to 2.5 million in 2012. This large-scale dairy conversion has been accompanied by increased irrigation, fertiliser application and the introduction of nitrogen-fixing plants (Quinn 2000).

Not surprisingly, this large-scale land-use change has resulted in marked changes to waterways. Stream channelisation and wetland drainage has been extensively undertaken in agricultural regions (Collier et al. 1995; Quinn 2000), such that many waterways in lowland regions are now modified drains. These drains are often considered to have poor biological diversity and little ecological value. In contrast with this perception, agricultural drains can provide habitat, support invertebrate and fish species and are often the last remnants of substantial wetlands that historically covered New Zealand's lowland areas (Young et al. 2004; James 2011).

Pastoral development has had profound impacts on water quality, aquatic habitats and invertebrate and fish communities . The clearing of stream bank vegetation reduces organic matter entering the stream, and loss of shading results in an increase in water temperature, nuisance plant growth and altered dissolved oxygen regimes (Quinn et al. 1997; Rutherford et al. 1997; Quinn 2000; Scarsbrook et al. 2001). Increased run-off, stock trampling and erosion cause higher levels of suspended sediment and turbidity (Trimble and Mendel 1995; Nguyen et al. 1998). Furthermore, higher dissolved nitrogen and phosphorus, and faecal indicator bacteria concentrations are typically found in pasture streams compared to those flowing through native forest (Quinn et al. 1997; Parkyn and Wilcock 2004). Native fish communities are impacted by the loss of stream shade, lack of suitable spawning sites and increased sediment inputs (Parkyn and Wilcock 2004). Invertebrate communities are also affected by increased water temperatures, nutrient concentrations and increased algal biomass (Quinn and Hickey 1990; Parkyn and Wilcock 2004). Agricultural streams tend to have higher invertebrate abundances, but fewer species that either consume leaf litter or are sensitive to pollution (Quinn and Hickey 1990; Parkyn and Wilcock 2004). One study reported that in catchments where at least 30% of the catchment has been converted to agriculture, freshwater invertebrate communities shift from sensitive clean water taxa to pollution tolerant species (Storey and Cowley 1997). The majority of lowland rivers in New Zealand are now in poor condition due to the land use changes that have occurred within their catchments (Collier et al. 1995).

Aquatic macrophytes

Spread of introduced macrophytes

There have been significant changes in the aquatic flora of New Zealand streams since human settlement (Reeves et al. 2004). New Zealand native macrophyte species are low growing, forming shallow mats and are not vigorous competitors (Reeves et al. 2004). Widespread, rapid deforestation has resulted in increased light and sediment inputs reaching stream environments, facilitating the establishment of introduced aquatic macrophyte species (Lacoul and Freedman 2006). In New Zealand, several introduced macrophyte species have been particularly successful invaders, including *Egeria densa*, *Elodea canadensis, Lagarosiphon major*, *Ceratophyllum demesum, Erythranthe guttata* and *Nasturtium microphyllum*. Species introductions occurred both deliberately and accidentally, via a number of routes: imported with live fish to oxygenate waters (e.g. *E. canadensis*), in ship ballast, for stock grazing (e.g. reed sweet grass, *Glyceria maxima* as a feed crop in wet areas), as ornamental plants (*E.* guttata), as vegetable crops (notably watercress, *N. officinale* and *N. microphyllum*), or from the aquarium and pond trade (Champion et al. 2002). Human activities have accelerated their spread, with fragments and seeds transferred on boats, fishing nets, float planes and weed management equipment. As a result, native species have either been eliminated or excluded from their preferred habitats and introduced macrophytes now dominate many New Zealand lowland streams (Champion and Tanner 2000; Champion et al. 2002; Reeves et al. 2004). Champion et al. (2002) suggests part of their success has been attributed to the lack of native species able to successful occupy some of their niches. Today, legislation under the Biosecurity Act (1993) is designed to prevent the sale and distribution of invasive macrophyte species to avoid further spread (Wilcock et al. 1999; de Winton et al. 2009).

Early records suggest that several introduced macrophyte species were deliberately released into New Zealand (Champion et al. 2002). The first recorded species to be introduced was *Elodea canadensis* in 1868. Due to their comparatively recent invasion in New Zealand waterways, macrophytes have not yet reached some more isolated regions including Fiordland and part of Northland (de Winton et al. 2009). However, their absence seems to be a result of their limited dispersal ability, rather than any lack of suitable habitat. Therefore, it is likely that aquatic macrophytes have not yet spread to their full range and further spread is to be expected

(Clayton and Edwards 2006). The threat of further species introductions also remains, where three of the worst weed macrophytes internationally (eurasian watermilfoil *Myriophyllum spicatum*, water primrose *Ludwigia peruviana* and water chestnut *Trapa natans*) have not yet been reported in New Zealand (Champion et al. 2002).

Issues surrounding macrophytes in aquatic ecosystems

Aquatic macrophytes provide important services in freshwater ecosystems, providing habitat for invertebrates and fish, regulating flow conditions, cycling nutrients, producing dissolved oxygen and adding organic matter to the waterway (Dawson and Haslam 1983; Sand-Jensen and Mebus 1996; Collier et al. 1999; Fleming and Dibble 2014). However, introduced macrophytes can grow prolifically and negatively affect waterways by filling waterways, reducing water flow, increasing sediment deposition, impeding drainage and causing flooding of agricultural land, and altering the community structure and abundances of aquatic invertebrates and fish (Fox 1992; Wilcock et al. 1999; Champion and Tanner 2000; Bączyk et al. 2018).

Management of stream macrophytes

Historically, drainage has been the primary function of many small agricultural streams in lowland regions. To be effective, waterways must remove excess water efficiently and quickly (Hudson and Harding 2004; Greer et al. 2012). When excessive macrophyte growth occurs, particularly in spring and summer, it can reduce the flood capacity of the drain (Nikora et al. 2008). Consequently, weedy macrophyte control is common practice (Fox 1992). The three main management strategies used in small flowing waterways are mechanical clearance, chemical application and hand weeding. Biological control has also been used to a lesser extent (Wells et al. 2003; Hudson and Harding 2004). All management options have a degree of impact on the waterway; however, the costs and benefits are usually considered and the most appropriate methods selected for each specific situation (James 2011).

Mechanical clearance

Mechanical clearance is widespread across New Zealand, with approximately 15,500 km of waterway cleared annually by local government (Greer 2014). Mechanical clearance usually involves the use of a bank-based digger with scoop bucket to physically remove macrophyte biomass (James 2011). Its effectiveness tends to be short-lived as rapid regrowth of macrophytes often occurs after clearance. Furthermore, multiple clearances are often required during a single growing season (Dawson and Haslam 1983; Fox 1992; Young et al. 2004).

The regular disturbance created by mechanical clearance resets macrophyte succession and the increased availability of light and space likely contribute to increased macrophyte establishment in streams. In a Marlborough study, one month after mechanical clearance, macrophyte communities were beginning to re-grow, and after three months, the beds had re-established (Young et al. 2004). The practice also creates plant fragments, which can be distributed downstream by river flows and re-establish, allowing species to spread (Zehnsdorf et al. 2015). Frequently, spoil removed from the channel is dragged up the stream bank and dumped on top of the bank to decompose. As a result, banks get built-up and the channel deepens over time, which contributes to long-term bank erosion (Figure 1.1). A regular mechanical clearance regime selects for macrophyte species that grow rapidly, have high dispersal capability and short life cycles (Franklin et al. 2008). Over several successive years of mechanical clearance, pronounced negative impacts on macrophyte community diversity have been shown, as slower growing and poorly adapted species are outcompeted (Franklin et al. 2008; Zehnsdorf et al. 2015).

The practice of mechanical clearance is potentially damaging to the freshwater ecosystems. Macrophytes provide habitat for fish and invertebrate species. Aquatic fauna living in or on macrophyte beds can become trapped in plant material and are unintentionally removed from the waterway in the digger bucket (James 2011). Shortfin eels (*Anguilla australis*) are particularly vulnerable to removal, given their strategy to burrow into sediment when disturbed (Young et al. 2004). When spoil is dumped on the stream bank, eels tend to head downhill out of the macrophyte/sediment in search of water. However, the likelihood of them heading in the right direction is greatly reduced as the spoil tends to slope away from the waterway and the risk of drying is rapid during warm summer months. Eels staying in the spoil ultimately leads to death by dehydration, suffocation and becoming stranded in the drying sediment. For example, Young et al. (2004) estimated that 0.3-0.4 eels were removed per metre of drain length mechanically cleared. In that study, invertebrate densities were found to be reduced by 50 % one week after mechanical macrophyte clearance, but recovery to pre-clearance levels occurred within two months.

The physical removal of macrophytes results in the re-suspension and mobilisation of sediment in the waterway (Figure 1.1) (Young et al. 2004; James 2011). The instream habitat and food sources for fauna remaining in the stream post-clearance is significantly reduced, impacting on their behaviour and life cycles with the potential for effects further along the food chain (James 2011). Following removal, macrophytes are usually left on the stream bank to decompose. This organic material may reenter the waterway through flooding or surface runoff and nutrients and sediment may return to the stream (Ministry of Agriculture and Forestry 2001; James 2011). A number of best management mitigation strategies have been proposed to minimise these effects, for example retaining refuges by only partially clearing a waterway, returning stranded fauna to the waterway, cleaning machinery between sites to prevent spread, avoiding harvesting during fish spawning seasons, and correct digger bucket selection based on site characteristics (Barrett et al. 1999; Madsen 2000; Ministry of Agriculture and Forestry 2001; James 2011; Greer et al. 2012).

Figure 1.1. Effects of mechanical macrophyte clearance on a small stream system. **A**, Digger removing macrophyte biomass. **B**, Excavator scoop bucket, showing macrophyte removal and release of suspended sediment. **C** and **D**, Stream reaches after mechanical clearance showing macrophyte biomass dragged up stream bank and placed to create over-steepened banks.

Chemical control

Chemical control involves the spraying of herbicide to kill macrophytes *in situ*. Herbicide may be applied from bank-based human-operated backpacks through to spraying from a vehicle, boat or helicopter depending on the scale of the operation. Two herbicides, diquat dibromide and endothall, are registered for use on submerged macrophytes in New Zealand (James 2011). A third herbicide, glyphosate is a non-selective, broad-spectrum herbicide commonly used on emergent and marginal macrophytes (Fox 1992; de Winton et al. 2013). Glyphosate is one of the world's most effective and most frequently used herbicides. It is absorbed through leaves and is transported by the plant to the growing points in roots and shoots, where it prevents the plant from being able to synthesise proteins that are required for growth (Magbanua et al. 2013). Following manufacturer's instructions, spraying directly on the waterway should be minimised due to the potential adverse toxicity effects on aquatic organisms (Hudson and

Harding 2004); however, investigations undertaken by Folmar et al. (1979) suggested glyphosate application at recommended rates on emergent marginal vegetation should not affect fish or macroinvertebrates.

Depending on the region of New Zealand where spraying is taking place, consent may be required from the relevant regional council (Young et al. 2004). All three herbicides have a Haznote classification of “toxic” or “very toxic” to aquatic organisms in their undiluted form, however they are considered to be non-toxic when applied at their recommended application rates. Chemical control has very little initial impact on macrophyte biomass, where after spraying it may take days or weeks for plants to die back (James 2011). Following spraying, herbicides can take several weeks for plants to die (de Winton et al. 2013) and the effectiveness is relatively short lived as plants can recover within several months. Young et al. (2004) found that after spraying with diquat, target plants were dessicated after one week, and one to three months after spraying die back was evident and plants had collapsed in the stream. Six months after spraying, plant cover had recovered to levels comparable to those prior to treatment.

Chemical control is not without its own range of adverse effects. The main concern surrounds the toxicity of the herbicide in the aquatic environment. This is complex to test, as it relies on the safety tolerance between the specified application rate and the level upon which it becomes hazardous/toxic (James 2011). Generally, once diluted, concentrations of herbicides used for macrophyte control are well below levels likely to harm aquatic fauna (Brooker and Edwards 1975). However, the testing regime for maximum allowable concentrations of specified herbicides fails to take into consideration the interaction between multiple stressors, which can result in the under or over estimation of risk (Kelly et al. 2010). In an investigation of the combined and independent effects of exposure to glyphosate and the parasite *Telogaster opisthorchis* on the development of spinal malformations in the fish *Galaxias anomalus*, Kelly et al (2010) found fish exposed to parasites developed spinal malformations, and deformations were more severe when exposed to both parasites and glyphosate. The effectiveness of herbicide application is reduced by rainfall within a specified period and on windy days spray drift can occur. Some macrophyte species are resistant to specific herbicides and non-selective herbicides can harm non-target plants, such as in adjacent areas of riparian plantings (de Winton et al. 2013).

The decomposition of dead macrophytes left in the channel after chemical control can potentially deplete oxygen to levels that may harm aquatic fauna (Jewell 1971; Brooker and Edwards 1975; James 2011). However, few studies have quantified oxygen concentrations before and after herbicide application. One study found that after diquat application, oxygen consumption frequently exceeded oxygen production, as macrophytes were no longer photosynthesizing and began to rot (Strange and Schreck 1976). Other studies of the effects of herbicide application have shown no effect of low dissolved oxygen concentrations on aquatic life (Brooker and Edwards 1973; Young et al. 2004). Furthermore, the decay of plant material can result in the rapid release of nutrients, especially phosphorus (James 2011). This high nutrient environment stabilized by decaying plant material forms an ideal habitat for macrophytes to regrow. The sudden change in habitat from macrophyte stands to decomposing plant material can drastically affect habitat and food sources for aquatic fauna, especially where invertebrate species are lost completely (Young et al. 2004).

The use of herbicides has been receiving increased public interest and concerns have been raised about the effects of glyphosate on both human health and aquatic life. In Canterbury, public pressure has resulted in the Christchurch City Council committing to limiting the use of glyphosate-based sprays to areas with no public access or where there are no other suitable alternatives. Pressure is mounting on other local and national government agencies to follow suit.

Hand weeding

Hand weeding involves the manual physical removal of macrophyte biomass using a sickle or scythe. This method is a very selective control option, where nuisance plants can be removed while desirable plants are left intact, but is a very labour intensive and costly macrophyte control technique and plants can usually re-grow quickly unless root material is also removed (de Winton et al. 2013; Bellaud 2014; Hussner et al. 2017).

Biological control

Biological control involves the release of new organisms (insects, fish or microorganisms) to feed upon macrophyte stems, leaves and roots. Importation, development, field testing and release of new organisms in New Zealand are tightly controlled by the Environmental

Protection Agency due to the potential risk they pose. In New Zealand, insect releases have been limited to a flea beetle (*Agasicles hygrophila*) and a moth (*Arcola malloi*) that target alligator weed. These two species selectively only target alligator weed, and are limited by temperature to the upper North Island.

Grass carp (*Ctenopharyngodon idella*) are herbivorous fishes, that were first introduced in 1971 and are considered unlikely to naturalise in New Zealand (Clayton and Wells 1999; Hofstra 2014; Hofstra et al. 2014). They are unselective feeders and will provide total control of submerged vegetation. Containment is an issue, and grass carp must be confined to the target area (Wells et al. 2003). In the Waikato region, grass carp have been introduced to several areas and have been successful in controlling target macrophytes. Wells et al. (2003) found that two months after release, macrophyte cover was significantly reduced. However, releases in New Zealand agricultural drains have largely been unsuccessful. Fish survival is dependent on water temperature, pH, sufficient dissolved oxygen levels and adequate water quality. Grass carp have been released at least three times in Simpsons drain, Hauraki, with no survival due to the low dissolved oxygen and low pH (Wells et al. 2003). Grass carp tend to avoid shallow water (less than 1 m deep) (Wells et al. 2003). They may be ineffective in the South Island, as they cease feeding at temperatures below 13^{o}C (Rowe and Schipper 1985; Hudson and Harding 2004). Removal of fish after macrophytes have been controlled is also problematic (Champion et al. 2002). For these reasons, biological control of macrophytes is not further investigated or discussed in this thesis.

Summary of macrophyte management techniques used in small agricultural streams

Mechanical clearance, chemical control and hand weeding are the most commonly used macrophyte control techniques used in small agricultural streams in New Zealand. Given these methods can be short lived, disruptive or high cost (Table 1.1), research is being undertaken to investigate other less common management techniques. Fundamental questions exist surrounding the efficacy of alternative techniques, however, there is the potential to offer more effective, cheaper, long-term and less ecologically damaging alternatives to traditional macrophyte control.

Table 1.1: Summary of three commonly used macrophyte management techniques used in small agricultural streams in New Zealand

	Mechanical control	**Chemical control**	**Hand weeding**
Description	Digger used to excavate macrophyte biomass	Herbicide sprayed to kill macrophytes *in situ*	Manual physical removal of plant biomass
Impact on aquatic biota	Fauna can become entrained and removed in extracted plant material Habitat removed	Potential direct toxicity effect on fauna Habitat and food sources altered	Minimal; fauna unharmed other than removal of pest species
Issues surrounding technique	Sediment resuspended and mobilised	Macrophytes left in drain to decompose Potential for oxygen to be depleted to harmful levels	Labour and cost intensive Care is required to remove all plant material including roots for longer-lasting results
Timing of results/ effectiveness	Immediately effective	Can take days or weeks for plants to die back	Immediately effective
Length of result	Effectiveness is short lived; recovery within 3 months Multiple clearances may be required in a growing season	Effectiveness is short lived; recovery to original levels within 6 months Multiple applications may be required in a growing season	Effectiveness short lived; ongoing maintenance required
Cost	$2,500 / hectare*	$290 / hectare*	$20,000 / hectare*
Disposal of material	Usually left on bank to decompose – can re-enter waterway	Left in channel to decompose, potentially depleting oxygen levels	Needs to be disposed of

* Cost taken from de Winton et al. (2013)

Thesis structure and chapter outlines

I carried out the research for my PhD as part of the Canterbury Waterway Rehabilitation Experiment (CAREX, www.carex.org.nz) a project undertaken within the Freshwater Ecology Research Group at the University of Canterbury. This long-term, collaborative project involved working with stakeholders to evaluate tools to achieve freshwater restoration success in agricultural streams. The project is focussed on nine small lowland agricultural waterways on the Canterbury Plains of the South Island of New Zealand and built strong partnerships with landowners, stakeholders and local management agencies. CAREX aimed to develop practical solutions to target multiple stressors, including excessive aquatic macrophytes, high nutrients, high deposited fine sediment and low aquatic biodiversity. My research focussed on understanding the factors which might limit excessive nuisance macrophytes and trialling practical alternative tools to control macrophytes in small lowland agricultural waterways.

My thesis has been written as a series of three stand-alone scientific papers for publication. Given this, there is some repetition in material presented in the introduction and methods sections. While all chapters were multi-authored, the fieldwork, writing and statistical analysis were primarily my own. The co-authors assisted with study design, fieldwork, provided advice on data analysis and commented on drafts of the manuscripts. In the chapters, I refer to “we”, since chapters were multi-authored work that have or will be submitted for publication. Tables and figures are included in each chapter, with one numbering system running throughout the thesis. Supplementary material is found at the end of each chapter. All references are collated into a single list at the end, to avoid repetition.

In Chapter 2, I undertook a region wide survey of 28 small streams to understand factors that influence macrophyte species diversity and percent cover. I proposed and tested a conceptual model of factors influencing macrophyte biomass and the scales at which they operate. This chapter has been submitted to Aquatic Botany and is currently under review: *Collins KE, Febria CM, Warburton HJ, Devlin HS, Goeller BC, McIntosh AR, Harding JS. 2018 submitted. Understanding factors that influence macrophyte diversity and abundance in New Zealand agricultural streams to inform macrophyte management. Aquatic Botany.*

In Chapter 3, I undertook two small-scale experiments testing practical macrophyte control tools. Firstly, I tested the effectiveness of alternative tools, including physical disturbance, flower and seed removal, intensive hand weeding, herbicide spray, partial shading, sediment removal and weed mat in 4 m^2 plots in an agricultural waterway. Secondly, I tested the impact of shading providing 80 % light reduction across the entire stream channel on macrophyte growth by constructing three 5 m shade tunnels. This chapter has been published in a peer-reviewed scientific journal: *Collins KE, Febria CM, Warburton HJ, Devlin HS, Hogsden KL, Goeller BC, McIntosh AR, Harding JS. 2018. Evaluating practical macrophyte control tools on small agricultural waterways in Canterbury, New Zealand. New Zealand Journal of Marine and Freshwater Research. DOI: 10.1080/00288330.2018.1487454.*

In Chapter 4, I undertook two further experiments scaling-up successful treatments from Chapter 3. Firstly, a 50 m reach-scale trial of intensive hand weeding, polythene shading and weed mat, and secondly, a large scale trial of 400 m weed mat and 200 m hand weeding. This chapter has been written as a paper but has not yet been submitted to a scientific journal for peer review.

In Chapter 5, I discuss the overall findings from Chapters 2 – 4, highlighting key results, discuss assumptions about my research, gaps in knowledge, future research directions and suggest how macrophyte management could be improved to aid the restoration of lowland agricultural streams.

Plate 2. A lowland agricultural roadside stream completely choked with excessive macrophyte growth dominated by *Erythranthe guttata*.

Chapter 2:
Understanding factors that influence macrophyte abundance in small New Zealand agricultural streams to inform macrophyte management

This chapter is formatted in the style of the journal Aquatic Botany:
Collins KE, Febria CM, Warburton HJ, Devlin HS, Goeller BC, McIntosh AR, Harding JS. 2018 submitted. Understanding factors that influence macrophyte diversity and abundance in New Zealand agricultural streams to inform macrophyte management.

Highlights

- Macrophyte diversity was low, with 13 species found across 28 agricultural streams
- The introduced sprawling emergent species *Erythranthe guttata* and *Nasturtium microphyllum* dominated
- At the reach scale, disturbance limited, but sediment cover enhanced macrophytes
- At the patch scale, shading and water velocity limited macrophyte cover
- Understanding the scale at which different factors operate can inform macrophyte management

Abstract

Aquatic macrophytes provide important functions in fresh waters, however, during summer, waterways can become choked with macrophytes, requiring management. We conducted a survey of 28 small streams across Canterbury, New Zealand, to determine factors that influence species diversity and percent cover at both the reach and patch scale. Overall, diversity was low with only thirteen species detected, and the introduced *Erythranthe guttata* and *Nasturtium microphyllum* dominated. Physical and chemical parameters, and macrophyte cover varied greatly both among stream reaches, and among patches within reaches. At the reach scale, a significant positive relationship was found between macrophyte and fine sediment cover, and significant negative relationships occurred between macrophyte cover and water temperature and dissolved oxygen. At the patch scale, significant positive relationships were recorded between macrophyte and fine sediment cover, sediment depth and distance to nearest riparian tree and significant negative relationships occurred between macrophyte cover and stream shade and water velocity. We present a conceptual model of factors influencing macrophyte growth indicating that the flow disturbance regime was the key factor limiting growth at the reach scale, compared to shading at the patch scale. Improving our understanding of the factors that influence macrophyte growth can inform alternative management regimes to manage excessive macrophyte growth.

Keywords: agricultural drains, plant management, aquatic weeds, disturbance regime, stream shade

Introduction

Globally, human activities including deforestation, irrigation, land-use conversion and intensification have altered ecosystem function and biodiversity in freshwater systems (Dudgeon et al. 2006; Vörösmarty et al. 2010). These actions have degraded environmental conditions, altered spatial connections in rivers and enabled the spread of pest plants and animals. In New Zealand, several introduced macrophyte species have been particularly successful invaders, including *Egeria densa, Elodea canadensis, Lagarosiphon major*, *Ceratophyllum demesum, Erythranthe guttata* and *Nasturtium microphyllum*. Their invasion success has been attributed to the lack of competitive native species occupying their preferred niches (Champion et al. 2002). The introduction of non-native macrophyte species into New Zealand has occurred both deliberately and accidentally via a number of routes: import with live fish to oxygenate waters (e.g. *E. canadensis*); as solid ballast in ships, for stock grazing (e.g. reed sweet grass, *Glyceria maxima* as a feed crop in wet areas); as ornamental plants (*E. guttata)*; as vegetable crops (notably watercress, *N. officinale* and *N. microphyllum*); and from the aquarium and pond trade (Champion et al. 2002). Furthermore, widespread deforestation and agricultural expansion have allowed introduced freshwater macrophyte species to establish in new ecosystems, while human activities have accelerated their spread in lakes and large rivers, with fragments and seeds transferred on boats, fishing nets, float planes and in smaller streams on weed management equipment (Champion et al. 2002). Consequently, it is common for introduced macrophytes to dominate modified lowland streams in New Zealand. Understanding the factors that influence macrophyte diversity and abundance in streams is imperative if we are to identify environmentally sustainable management tools.

Aquatic macrophytes are often present in streams, either rooted in the bed of soft-bottom streams, or in the bank or patches of fine sediment of hard-bottomed streams. They provide important services including: nutrient cycling, habitat and food for some benthic invertebrates and fish, and re-oxygenation of water (Dawson and Haslam 1983; Sand-Jensen and Mebus 1996; Collier et al. 1999; Fleming and Dibble 2014). In lowland streams, their distribution and abundance is likely affected by a number of environmental factors, and the importance of any particular factor varies with spatial and temporal scale. Various studies have suggested macrophytes are influenced by numerous abiotic factors such as: river disturbance, water velocity, light availability, bed substrate and sediment characteristics, turbidity, water depth, nutrient availability, and biotic factors including: competition between species, and grazing (Dawson and Haslam 1983; Canfield and Hoyer 1988; Fox 1992; Reeves et al. 2004; Young et al. 2004; Bowden et al. 2007; Franklin et al. 2008; James 2011; Cornacchia et al. 2018). We propose a conceptual model (Figure 2.1) to summarise these factors acting at different spatial scales and to provide a framework for investigating how nuisance macrophyte growth can be managed. In their review paper, Franklin et al. (2008) suggested water velocity was the most important characteristic affecting local scale distribution and plant growth, whereas based on a survey of 17 Florida streams, Canfield and Hoyer (1988) found shading provided by riparian vegetation was the most important factor controlling the distribution and abundance of macrophytes. Canfield and Hoyer (1988) demonstrated a strong inverse relationship between forest canopy coverage and macrophyte biomass and provide an equation to estimate macrophyte standing crop biomass based on percent canopy coverage.

Not only are macrophytes affected by external environmental conditions, they also have been described as “ecosystem engineers” because they alter water velocity, water depth and sediment dynamics (Fox 1992; Franklin et al. 2008; Booker and Snelder 2012; Cornacchia et al. 2018). Water velocity is decreased in macrophyte patches due to increased hydraulic roughness, resulting in deeper, slower flowing channels and facilitating the deposition of suspended sediment (Collier et al. 1999; Champion and Tanner 2000; Willis et al. 2017). This also creates ideal conditions for further macrophyte growth. Outside of macrophyte patches, water velocity increases, resulting in scour and increasing physical stress on the edges of macrophyte patches (Cornacchia et al. 2018). Thus, the positive feedbacks associated with macrophytes are important to investigate and understand.

During summer months, introduced macrophytes in lowland agricultural streams can proliferate and reduce water velocities, increase sediment deposition, impede drainage causing flooding of agricultural land, create large daily fluctuations in dissolved oxygen concentrations and alter invertebrate and fish communities (Fox 1992; Champion and Tanner 2000; Bączyk et al. 2018). Under these circumstances plant removal is commonly undertaken by landowners and water management agencies (Fox 1992). In many countries, including New Zealand, management of introduced macrophytes in streams traditionally involves mechanical clearance, herbicide spray or hand weeding to remove or control biomass. Mechanical clearance typically involves a bankside digger physically excavating macrophyte biomass (James 2011). Whereas, herbicide spray involves the application directly onto the plant; and hand weeding involves workers cutting off macrophyte stems using a sickle or scythe to physically remove the plants. These management practices can be both costly and destructive to in-stream habitat (Hudson and Harding 2004; James 2011) and can result in increased spread of macrophytes. Thus, understanding the drivers of macrophyte abundance and growth will be useful in identifying and testing alternative management tools.

We investigated drivers of diversity and abundance in agricultural streams in order to better inform the management and control of introduced macrophytes. Small, highly-modified, spring-fed streams with high nitrate-nitrogen concentrations (> 1 mg/L) were surveyed. We hypothesised that macrophyte diversity and abundance were controlled by drivers operating at different spatial scales: the reach scale and at a localised patch scale (Figure 2.1). Specifically, we predicted that highly physically stable sites would have high macrophyte abundance and that macrophytes would decrease as both stream shade and stream velocity increased. In our study region, we expected that nitrate-nitrogen concentrations would exceed levels likely to limit macrophyte growth, and therefore, phosphorus might become the limiting nutrient. We also predicted that macrophyte abundance would increase as stream bed sediment cover increased and, finally, that the effect of herbivory on macrophyte abundance would be minimal as there are no native herbivorous fish and very few invertebrate taxa that feed on macrophytes in New Zealand.

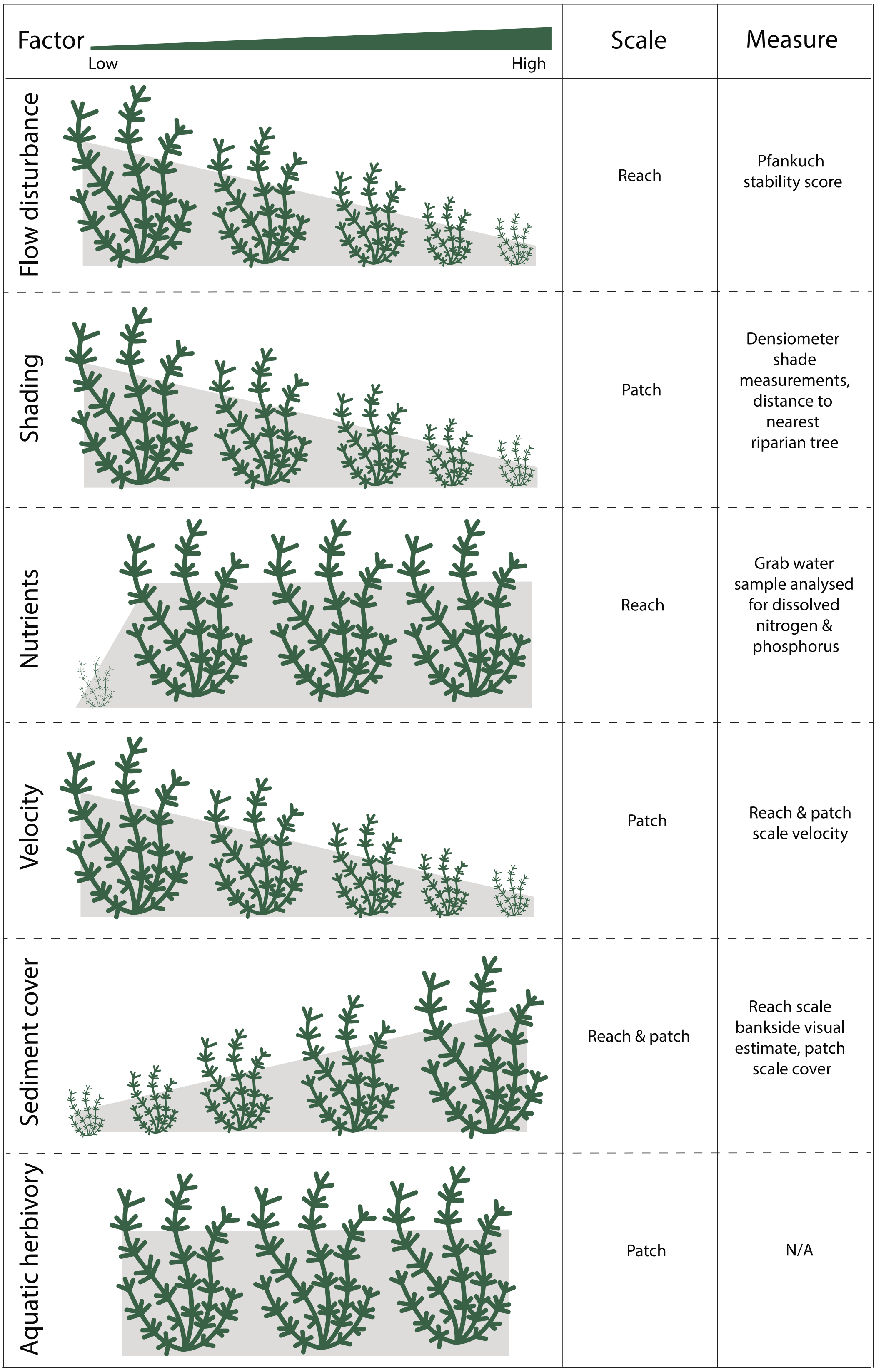


Figure 2.1. Conceptual model of factors likely to affect macrophyte abundance in small agricultural streams in New Zealand and the scale which they operate at. Factors increase from left to right along the x-axis, with larger plant sizes indicative of greater macrophyte abundance.

Materials and methods

Survey sites

We surveyed 28 small, 1st–2nd order streams (< 5 m wetted width) in the Canterbury Region, on the east coast of the South Island of New Zealand (Figure 2.2). The area included the lowland Canterbury Plains, volcanic Banks Peninsula and high-country foothills of the Southern Alps. Since the European settlement of Canterbury in 1850, the surrounding Canterbury Plains have been gradually and extensively modified (Pawson and Holland 2008). Wetlands were drained for pasture, waterways straightened and drains constructed to convey water effectively to the coast (Pawson and Holland 2008; Winterbourn 2008). Since 1990, large-scale land-use conversion and intensification has been undertaken, changing from sheep farming and cropping to intensive dairying (Scarsbrook et al. 2016; DairyNZ 2017). Intensive agriculture now dominates the expanse of the Canterbury Plains. In contrast, Banks Peninsula is a small volcanic protrusion off the eastern edge of the Canterbury Plains consisting of two volcanic calderas. At 920 m at its highest point, the Peninsula has nearly 100 small, high gradient and high velocity catchments (generally < 3rd order) that discharge directly to the sea (Winterbourn 2008). Streams in the high-country foothills of the Southern Alps originate from alpine rock or springs, and flow through scrub and tussock grassland before entering large braided rivers and crossing the pastoral lowlands before discharging to the Pacific Ocean. The foothill streams we investigated were on low-intensity sheep farms.

Streams were selected based on macrophyte data from the State of the Environment database collected by Environment Canterbury, the regional government agency monitoring streams (Booker and Snelder 2012). This database contained 8875 records from 182 sites collected between 2004 and 2014. We selected sites from this database with macrophytes present in 2014. Given we only surveyed sites that were known to have macrophytes present, some components of our conceptual model (Figure 2.1) were not tested (e.g. high flood disturbance sites were not included). Surveyed streams were not located downstream of other sampled sites and were all flowing through agricultural landscapes in open farm or tussock land. Streams were sampled on a single occasion at the height of the macrophyte growth season under austral summer flow conditions in February 2017. Sites sampled all had macrophytes present and represented a gradient of macrophyte stream bed cover from 5 % through to 100 %. Most agricultural stream reaches investigated in this study have had macrophyte control undertaken each year over the last several decades, however, the streams had not yet been managed during the season our study was undertaken, and we are confident that previous management actions have not impacted our results.

In each stream, assessments were made at two spatial scales; the reach (50-m) and the patch (30 x 30 cm quadrats) scale.

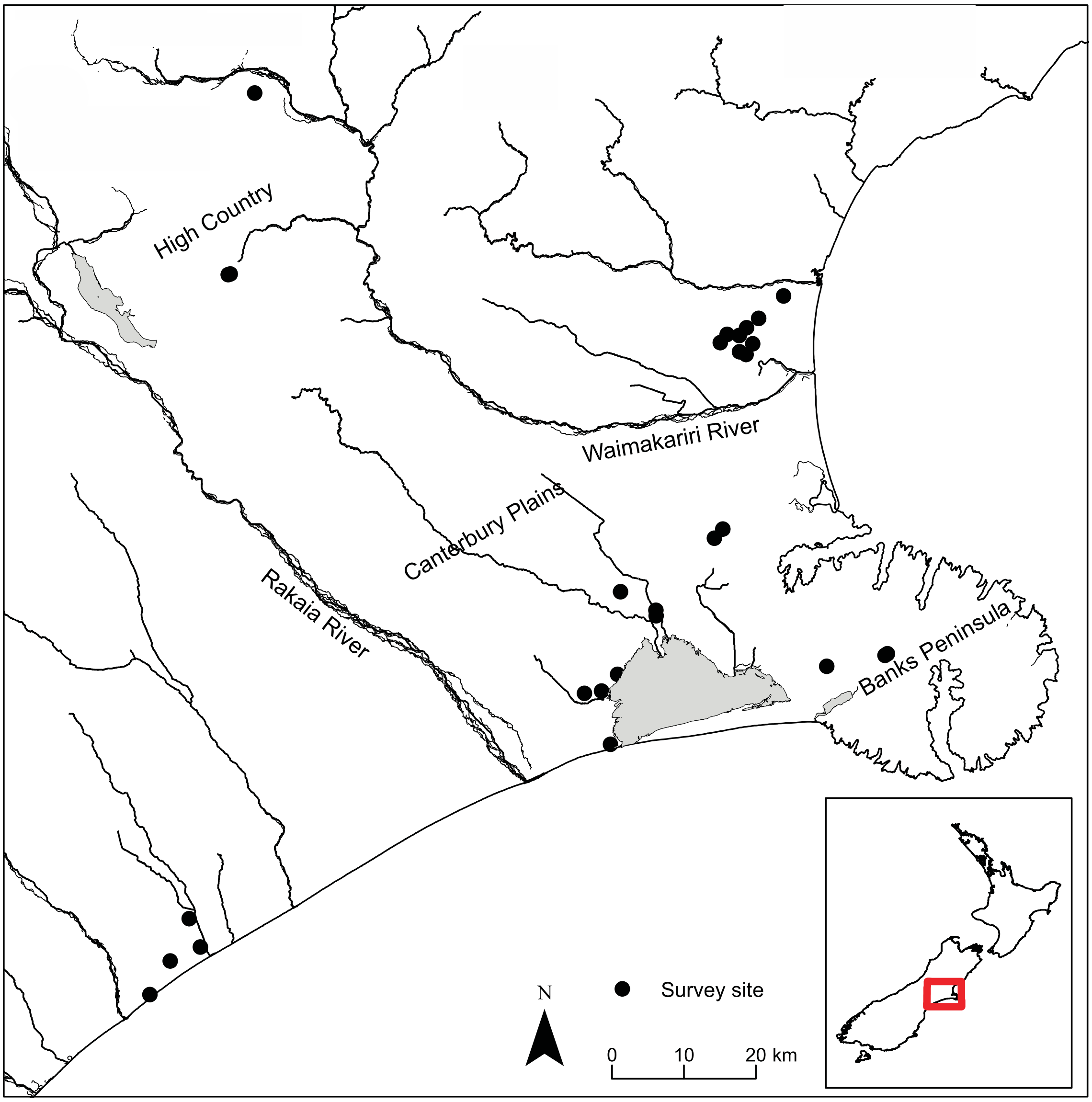


Figure 2.2. Twenty-eight first and second order streams surveyed for macrophytes in Canterbury (43.7542°S, 171.1637°E), South Island, New Zealand.

Reach scale

At each stream a 50 m reach was identified, which included a riffle, run, pool complex where possible. Macrophyte species present within the reach were recorded by walking the reach and identifying all species present. Visual bankside estimates of percentage macrophyte bed cover and percentage bed sediment cover were made for the reach following protocols of Harding et al. (2009). Bank stability, bed stability and flood disturbance were measured using the channel stream stability index of Pfankuch (1975). Scores for each physical variable are weighted then added together to give a combined assessment of the physical disturbance of the reach. The index ranges from 38 indicating a highly stable system, to 152 indicating a highly unstable system (Pfankuch 1975).

At the downstream end of each reach, spot water chemistry was recorded, including temperature, pH, specific conductivity (YSI Pro 1030, Yellow Springs, USA), dissolved oxygen (YSI EcoSense ODO 200, Yellow Springs, USA) and turbidity (Hach portable turbidimeter 2100P, Colorado, USA). Due to time and resource constraints, we were unable to undertake longer-term detailed measurements of parameters. Wetted width and depth were measured, and water velocity was recorded using a Flo-Mate 2000 (Marsh McBirney, USA). Measurements were made across a transect, and discharge (m^3 s^{-1}) was calculated using the area integration method (Harding et al. 2009; Gordon et al. 2012). A grab water sample was collected mid-channel, filtered (Whatman glass fibre fine 0.7 μm filters), and stored in an acid-washed (5% HCl) plastic bottle. Water samples were transported on ice, and frozen within 8-hours of collection. Samples were analysed for nitrate-nitrogen (cadmium reduction method; Rice et al. 2017) and dissolved reactive phosphorus (molybdate blue method; Rice et al. 2017) on an Easychem Plus analyser (Systea, Italy) at the University of Canterbury.

Patch scale

Within each 50-m reach, 10 – 20 quadrats (30 x 30 cm) were placed in a semi-random stratified arrangement and variables measured. The stratified sampling method and differing numbers of quadrats were used per reach to ensure that the variability of small-scale environmental variables was captured. A total of 455 quadrats were measured across the 28 sites. In each quadrat a range of factors were measured, including: macrophyte species; visual percentage macrophyte cover; fine sediment percentage cover; sediment depth; water depth; stream shade; bank slope; water velocity; distance to bank; distance to nearest riparian tree (as a proxy for riparian shading) and average substrate size. A bathyscope underwater viewer was used to view the stream bed where required. Sediment depth was measured centrally in the quadrat by pressing a ruler into the sediment until firm substrate was reached. Stream shade was measured using a convex spherical densiometer (model A; Lemmon 1957) directly above each quadrat (Harding et al. 2009). Bank slope of both stream banks were measured adjacent to where the quadrat was placed in stream to provide a further indication of the level of shading likely to be provided by the stream bank. Water velocity and water depth were also measured at the centre of the quadrat. Average substrate size was measured on the Wentworth scale for 10 pieces of substrate from within each quadrat, by measuring on their intermediate axis using a gravelometer (Harding et al. 2009).

Statistical analyses

We evaluated the relationship between macrophyte cover and explanatory variables at the reach and patch scale in separate analyses. At the reach scale, explanatory variables included nitrate-nitrogen, dissolved reactive phosphorus, Temperature, pH, conductivity, dissolved oxygen, turbidity, reach sediment cover, stream width and discharge. At the patch scale, explanatory variables included quadrat sediment cover, sediment depth, stream shade, bank slope, water depth, velocity, distance to bank, distance to nearest riparian tree and average substrate size. Given species diversity was found to be low and highly variable between patches, no further analysis was able to be undertaken on macrophyte diversity.

Predictors were examined for collinearity based on the variance inflation factor (VIF). No predictors had VIF values > 10, thus predictors were said to be independent (Craney and Surles 2002). Principal components analysis (PCA) of physical and chemical parameters was undertaken separately at the reach and patch scales using the ‘prcomp’ function in base R (R Core Team 2014).

Two models were constructed separately for reach and patch scale explanatory variables. To test for any significant variables affecting macrophyte growth, we included all explanatory variables in models (i.e. no model selection was undertaken). At the reach scale (n = 28), an additive linear model including all reach scale environmental variables was constructed for macrophyte cover (Table S2.1) using the ‘lm’ function in base R (R Core Team 2014). We calculated partial r^2 values for all significant relationships ($P < 0.05$) to identify the proportion of variance explained by the specific factor while accounting for the other factors in the model

(Nakagawa and Schielzeth 2013). This was undertaken using the ‘partial.R2’ function in package ‘asbio’ (Aho 2016). At the patch scale (n = 455), an additive linear model including all quadrat scale explanatory variables was constructed for macrophyte cover using the ‘lmer’ function in packages ‘lme4’ (Bates et al. 2015) and ‘lmertest’ (Kuznetsova et al. 2016). At the patch scale, reach was included as a random effect to account for nesting of patches within stream reaches (Table S2.1). For this model, partial r^2 values were calculated using the ‘r2beta’ function in package ‘r2glmm’ (Jaeger 2017).

All statistical analyses were undertaken using R statistical software version 3.1.2 (R Core Team 2014) and macrophyte percent cover values were normalised by arcsine square-root transformation prior to analysis (Zar 2009).

Results

Reach scale

Stream reach-scale macrophyte cover ranged from 5 – 100 % across the 28 reaches (Table 2.1). A total of 13 macrophyte species were observed, but reaches typically had between 2 – 5 species (Tables 2.1 & 2.2). One reach had only one species, and three reaches had seven species, the maximum number observed at any one reach. The most commonly occurring species were *E. guttata* (present at 27 of 28 reaches), and *Nasturtium microphyllum*; (present at 26 reaches), whereas *Ranunculus trichophyllus*, *Myriophyllum triphyllum* and *Juncus articulatus* were all found at two reaches and *Azolla rubra* was found at only one reach (Table 2, Figure 2.3).

Physical and chemical parameters were variable across the 28 reaches, indicating we were successful in sampling reaches along a large gradient of conditions for each variable (Figure 2.4, Table 2.1). The key factors included in our conceptual model had the following ranges: deposited sediment cover ranged from 0 – 100 %, nitrate-nitrogen from 0.02 – 14.35 mg/L, dissolved reactive phosphate from 1 – 42 μg/L and discharge from 0.01 – 0.37 m^3/s. All reaches surveyed had relatively stable beds, with Pfankuch index scores ranging from 55 – 75 (Table 2.1).

Table 2.1. Reaches scale characteristics of 28 stream reaches surveyed across the Canterbury Region.

Variable (units)	Minimum	Maximum	Median (± 1 SE)
Macrophyte cover (%)	5	100	50 (5.71)
Macrophyte species richness	1	7	3 (0.34)
Nitrate-nitrogen (mg/L)	0.02	14.35	1.01 (0.78)
Dissolved reactive phosphorus (μg/L)	1	42	4 (2)
Temperature (°C)	8.6	18.7	14.25 (0.4)
pH	6.4	8.0	7.1 (0.1)
Conductivity ($\mu s_{25°C}$/cm)	38.1	394.3	193.6 (16.45)
Dissolved oxygen (% saturation)	11.9	119.9	72.6 (4.4)
Turbidity (NTU)	0.3	2.4	0.8 (0.1)
Sediment cover (%)	0	100	90 (7.66)
Stream wetted width (m)	1.2	4.8	2.35 (0.14)
Discharge (m^3/s)	0.005	0.372	0.031 (0.02)
Pfankuch stability score	55	75	65 (1)

Table 2.2. Macrophyte species observed and their frequency across 28 stream reaches surveyed in Canterbury; * indicates a native species.

Species name	Common name	Number of sites	% of sites
Erythranthe guttata	Monkey musk	27	96
Nasturtium microphyllum	Watercress	26	93
Glyceria fluitans	Floating sweetgrass	11	39
Myosotis sp.	Water forget-me-not	10	36
Veronica anagallis-aquatica	Water speedwell	7	25
Potamogeton crispus	Curly pondweed	5	18
Myriophyllum aquaticum	Parrot's feather	4	14
Elodea canadensis	Canadian pondweed	4	14
Callitriche stagnalis	Water starwort	3	11
Ranunculus trichophyllus	Water buttercup	2	7
Myriophyllum triphyllum	Water milfoil*	2	7
Juncus articulatus	Jointed rush	2	7

Azolla rubra	Azolla*	1	4








Figure 2.3. Stream reaches varied in width, shading, macrophyte cover and substrate size.

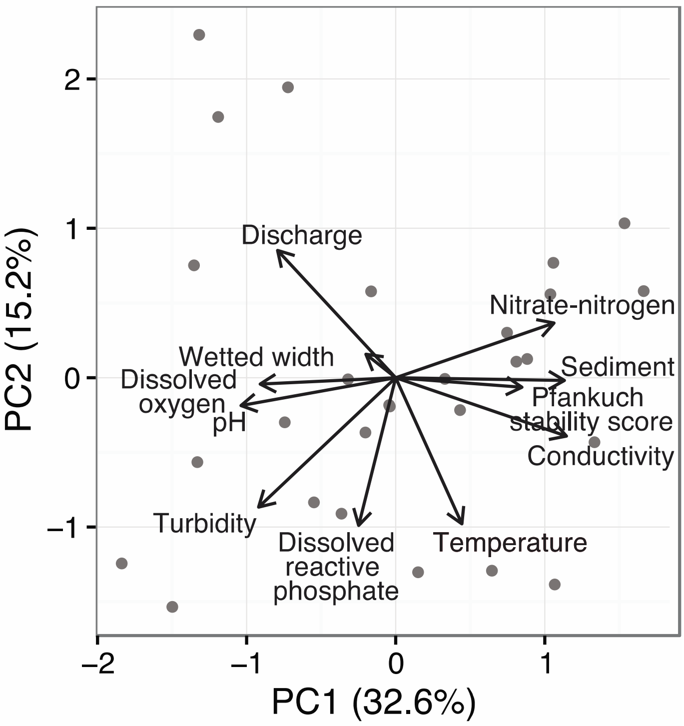


Figure 2.4. Principal components analysis (PCA) of physical and chemical parameters across 28 stream reaches in the Canterbury region, South Island, New Zealand. Each PCA axis represents an environmental gradient, with percent variation explained by each axis in brackets; the direction and length of the arrows denote the direction and strength of correlation. Closed circles indicate each individual reach.

Linear modelling showed a significant positive relationship between macrophyte cover and fine sediment cover ($t_{1, 17} = 2.35$, $P < 0.05$, $r^2 = 0.24$) (Figure 2.5, Table S2.1). In contrast, there were significant negative relationships between macrophyte cover and both water temperature ($t_{1, 17} = -2.16$, $P < 0.05$, $r^2 = 0.22$) and dissolved oxygen saturation ($t_{1, 17} = -2.58$, $P < 0.05$, $r^2 = 0.28$) (Figure 2.5, Table S2.1). Finally, there were no significant relationships between macrophyte cover and other measured variables including nitrate-nitrogen, phosphorus, pH, specific conductivity, turbidity, stream width and discharge (Figure 2.5, Table S2.1).

Because sites were surveyed on a single occasion at different times of the day, dissolved oxygen could be affected by the time of the day the sample was taken. However, we tested for this and found no effect of the time of day on dissolved oxygen saturation ($F_{1, 26} = 2.31$, $P = 0.14$). We also might expect an effect of altitude and dissolved oxygen saturation or water temperature. We found altitude of the site was not related to dissolved oxygen saturation ($F_{1, 26} = 0.24$, $P = 0.63$), however, there was a statistically significant negative effect of the altitude on water temperature ($F_{1, 26} = 37.2$, $P < 0.001$). Our 25 sites under 100 m a.s.l. had water temperatures between 12.5 – 18.7 ºC whereas the remaining 3 high country sites (between 550 and 800 m a.s.l) had temperatures between 8.6 and 12.1 ºC.

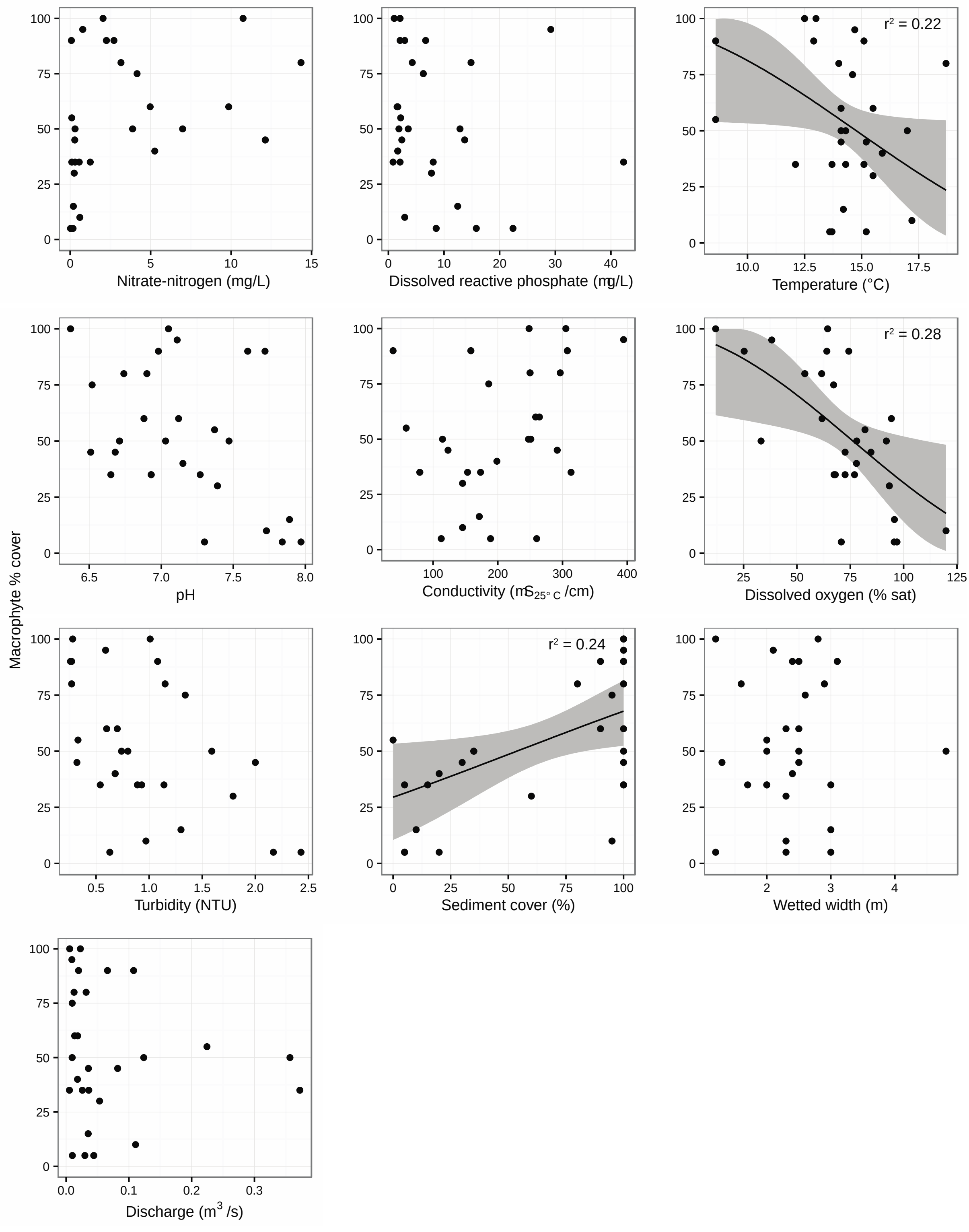


Figure 2.5. Linear mixed effects modelling of macrophyte cover and stream reach scale physical and chemical variables at the reach scale (n = 28). Points show raw data, with significant relationships (P < 0.05) indicated with the line of model fit ± 95 % confidence intervals (grey shaded area) and partial r^2 values. Macrophyte percent cover values were normalised by arcsine square-root transformation prior to analysis and back transformed to original scale for graphing.

Patch scale

As expected, there was a high degree of variability in physical and chemical parameters between stream patches (Figure 2.6). Macrophyte cover showed significant positive relationships with sediment cover ($t_{1, 288}$= 2.10, P < 0.05, r^2 = 0.02), sediment depth ($t_{1, 288}$ = 2.81, P < 0.01, r^2 = 0.03) and distance to the nearest riparian plant ($t_{1, 288}$ = 2.64, P < 0.001, r^2 = 0.03) (Figure 2.7, Table S2.1). There were significant negative relationships between macrophyte cover and stream shade ($t_{1, 288}$ = -7.55, P < 0.001, r^2 = 0.18) and water velocity ($t_{1, 288}$ = -5.08, P < 0.001, r^2 = 0.08) (Figure 2.7, Table S2.1). However, there were no significant relationships between macrophyte cover and other measured variables including bank slope, water depth, distance to bank and average substrate size (Figure 2.7, Table S2.1).

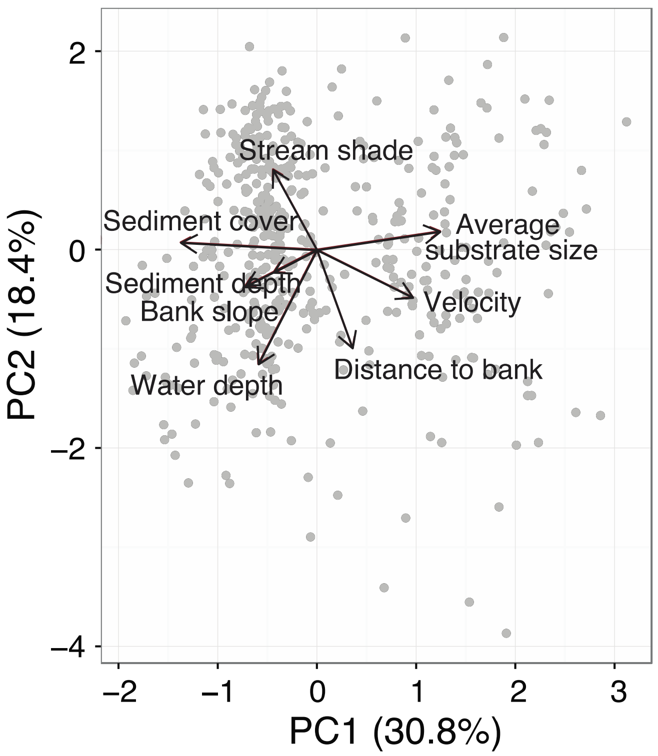


Figure 2.6. Principal components analysis (PCA) of physical and chemical parameters at the patch scale across 455 quadrats in the Canterbury region, South Island, New Zealand. Each PCA axis represents an environmental gradient, with percent variation explained by each axis in brackets; the direction and length of the arrows denote the direction and strength of correlation. Closed circles indicate each individual quadrat.

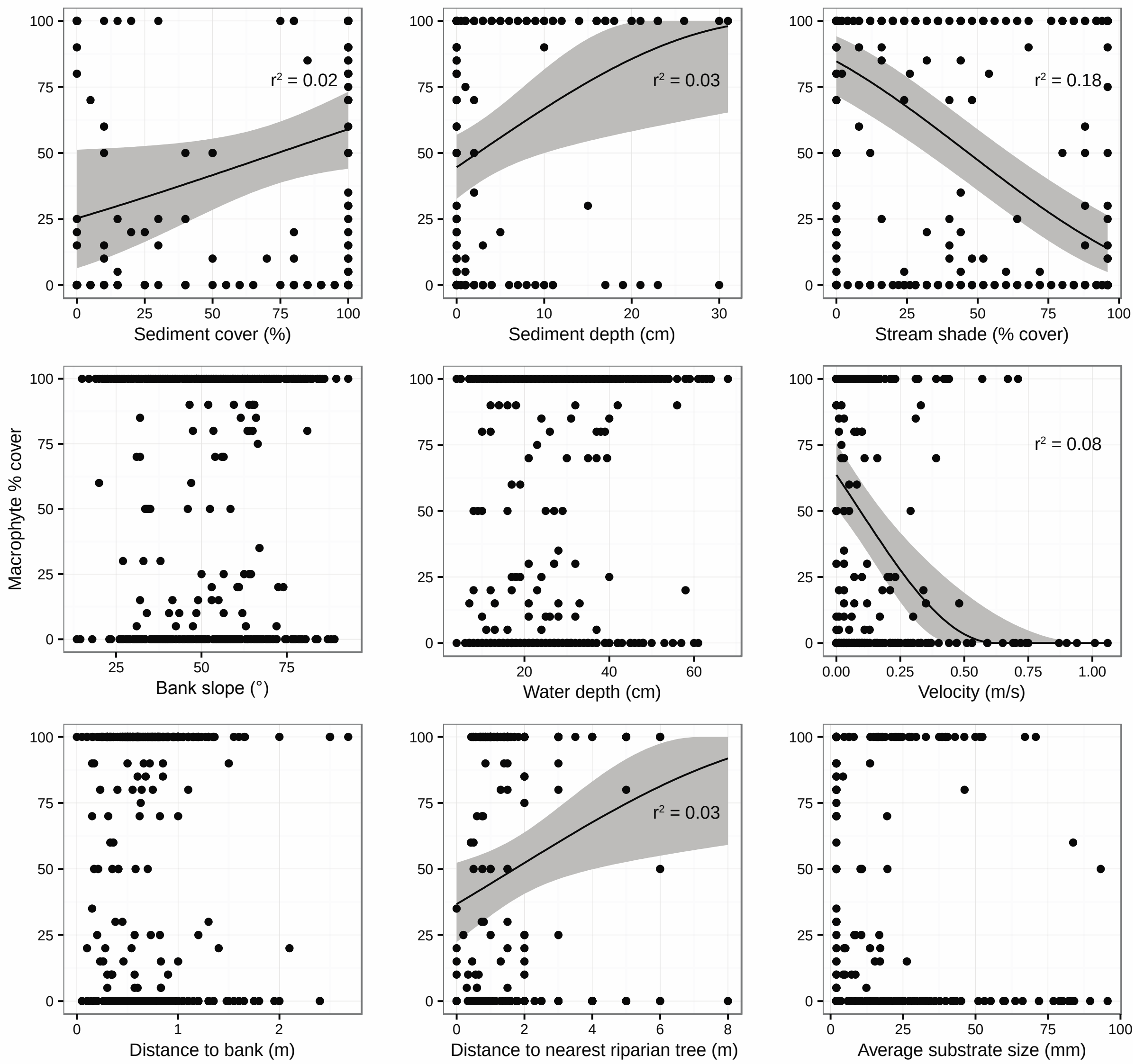


Figure 2.7. Linear mixed effects modelling of macrophyte cover and patch scale physical and chemical variables (n = 455). Points show raw data, with significant relationships ($P < 0.05$) indicated with the line of model fit ± 95 % confidence intervals (grey shaded area) and partial r^2 values. Macrophyte percent cover values were normalised by arcsine square-root transformation prior to analysis and back transformed to original scale for graphing.

Discussion

Our aim was to determine the physical and chemical factors that are associated with and influence macrophyte diversity and abundance.

Generally, stream reaches had low macrophyte diversity and only two out of the 13 species we recorded were native species. Most agricultural stream reaches in this study, particularly those on the Canterbury Plains, are highly modified and have been actively managed by local water management authorities over several decades. This removal of excessive macrophyte growth causes significant physical disturbance to the system and results in changes to stream morphology. On-going management of stream and river macrophytes via chemical control and mechanical excavation are recognised internationally as approaches that do not allow slower-growing native species to recover and often these species are lost, resulting in overall lower macrophyte diversity (Franklin et al. 2008; Zehnsdorf et al. 2015). In contrast, introduced species such as *E. guttata* and *N. microphyllum* that are fast growing, have high dispersal capability and short life cycles are able to thrive in these environments (Franklin et al. 2008). Anecdotally, we have observed some small unmanaged agricultural streams to have diverse macrophyte communities. Therefore the low macrophyte diversity in these stream reaches is likely an artefact of the management regime, and the high rate of anthropogenic bed disturbance.

In general, we were able to confirm most of the relationships in our hypothesised conceptual model (Figure 1). Given we only surveyed sites that were known to have macrophytes present, some parts of our conceptual model were not tested e.g. disturbed (flood-prone) streams. Also we did not evaluate the impact of aquatic herbivory on macrophyte abundance, as in New Zealand any effect is likely to be minimal (Matheson et al. 2012). Macrophyte herbivory occurs at low levels by a limited group of taxa, including aquatic insects, waterfowl and fish (Lodge 1991). Kōura (freshwater crayfish) can consume macrophytes, however they are also uncommon in Canterbury streams. There are very few other invertebrate macrophyte grazers in these systems. Furthermore, there are no native herbivorous fish or aquatic/semi-aquatic mammals in New Zealand, and introduced herbivorous fish including rudd (*Scardinius erythrophthalamus*) and grass carp (*Ctenopharyngodon idella*) are not common in Canterbury streams. Finally, aquatic birds (such as ducks, swans and Canadian geese) are also largely absent from small lowland streams. Hence, we believe any effect of herbivory on macrophyte abundance to be minimal in these systems and we were not able to examine its effects in this study.

Stream reach scale

Most of our stream reaches were categorised as physically stable and not subject to regular severe flooding, droughts or natural disturbance. This was to be expected, as a prerequisite for site selection was that macrophytes were known to be present from previous monitoring. Therefore, we were not able to adequately test the natural disturbance element of our conceptual model; however, we suggest natural disturbance is a key factor controlling the presence or absence of macrophytes. Riis and Biggs (2003) surveyed 15 streams on the South Island of New Zealand and found macrophyte cover and species richness decreased in streams where floods were more frequent. In their study, macrophytes were not found at sites with more than 13 annual events where flow was seven times greater than the median. However, some species (including *E. canadensis*, *Lagorosiphon major*, and *Potamogeton crispus*) are adapted with extensive root systems and high rates of propagule production to enable survival in high frequency flooding conditions (Riis and Biggs 2003). However, low flows and stream drying have also been shown to have a significant negative effect on macrophyte communities (Franklin et al. 2008).

Somewhat surprisingly we did not find any relationship between macrophyte cover and nitrate-nitrogen or dissolved reactive phosphorus concentrations. Nitrogen and phosphorus are the primary nutrients required for macrophyte growth, and their concentrations can limit plant establishment and maturation (Lacoul and Freedman 2006; Bornette and Puijalon 2011). However, flowing water constantly replenishes these nutrients; and several workers have suggested they may not be limiting factors controlling abundance (Fox 1992; Bowden et al. 2007). Further, most macrophyte species have a wide range of tolerance to nutrients and few species are limited by a lack or excess of nutrients (Janauer 2001). Nitrogen and phosphorus are taken up from the water column by both plant roots and leaves (Bristow and Whitcombe 1971; Madsen and Cedergreen 2002). Additionally, rooted macrophytes can take up nutrients directly from stream bed sediment (Franklin et al. 2008; Bornette and Puijalon 2011). From a study in Florida, Canfield and Hoyer (1988) calculated that macrophytes were only taking up <2% of the annual nutrient exports.Lowland agricultural streams are often saturated with nutrients; for example we recorded nitrate-nitrogen concentrations in excess of 14 mg/L. In agricultural streams in Canterbury, macrophyte beds dominated by *E. canadensis, J. articulatus, M. guttatus, N. microphyllum, P. cheesmanii,* and *R. trichophyllus* did not take up significant levels of nutrients from the water column, and were likely obtaining nutrients from sediments (O'Brien et al. 2014). This is supported by further work we have undertaken, suggesting that macrophytes in these systems root and establish on the banks of the stream, and extend growth out across the water surface (Collins et al. 2018a). This would seem to be particularly important for monkey musk, *E. guttata,* the most dominant species in our survey.

As proposed in our conceptual model (Figure 2.1), macrophyte cover increased as sediment cover and sediment depth increased at both the reach and patch scales. Stream bed substrate plays a key role in enabling macrophytes to establish and take up nutrients (Lacoul and Freedman 2006). Most rooted macrophyte species prefer environments made up of highly cohesive, fine inorganic sediment (Bornette and Puijalon 2011; Vukov et al. 2017). Furthermore, the presence of macrophytes in the channel decreases water velocity, resulting in entrained sediment becoming deposited and reinforcing the high sediment-macrophyte relationship (Collier et al. 1999; Champion and Tanner 2000; Willis et al. 2017).

Another finding was that as macrophyte cover decreased, water temperatures increased. In-stream shading can be provided by macrophytes themselves by reducing the direct solar radiation able to reach the water surface and regulating water temperature (Willis et al. 2017). Additionally, riparian shading can control water temperatures between 1 – 3 ^{o}C (Kristensen et al. 2013; Ryan et al. 2013; O'Briain et al. 2017), so locations with well-established planted riparian zones providing stream shade would be expected to have reduced water temperatures along with decreased macrophyte cover.

As macrophyte cover increased, dissolved oxygen saturation decreased. Low levels of dissolved oxygen can occur in dense macrophyte beds, especially of emergent or floating-leaved species (Frodge et al. 1990; Caraco and Cole 2002). These species can release much of the oxygen they produce through photosynthesis directly to the air rather than into the water column (Pokorný and Rejmánková 1983). Additionally, the high macrophyte abundance creates low water velocities which might result in less dissolved oxygen and carbon dioxide in the water column (Caraco and Cole 2002). Finally, the stream reaches with high macrophyte cover also had high bed sediment cover, which likely contain microbes that consume oxygen in the process of breaking down organic material.

Patch scale

Macrophyte cover decreased as stream shading increased at the patch scale, as proposed in our conceptual model (Figure 2.1). Light availability is another key factor determining the presence and abundance of macrophytes, affecting the ability of macrophytes to photosynthesise, thus limiting their distribution, growth and development (Dawson and Haslam 1983). We visually observed macrophytes being limited in shaded environments. Light availability at the water's surface is limited by day length, season and shading due to stream orientation, riparian canopy and stream banks (Matheson et al. 2012). In small agricultural streams, banks can become over-steepened by mechanical clearance and spoil deposited on the bank creates a bund, making banks higher. These narrow, incised streams have increased shading caused by the topography of the stream bank (Rutherford et al. 1997). However, riparian planting is more effective at providing stream shading in small to medium sized channels (stream order 1^{st}-3^{rd}), with decreasing influence as streams become wider (Poole and Berman 2001).

Our results indicated that macrophyte cover decreased as water velocity increased at the patch scale, with no macrophytes found where velocity exceeded 0.75 m/s. Water velocities of around 0.3 – 0.4 m/s have been shown to support relatively large biomass and species richness of macrophytes (Lacoul and Freedman 2006). Macrophytes are generally not found where velocities are greater than 1 m/s as plants are uprooted or suffer stem breakages and are swept away (Chambers et al. 1991). This aligns with our findings and suggests that stable streams with low velocities are more suitable for macrophyte establishment and growth (Riis and Biggs 2003).

There is an interaction between water velocity, high macrophyte abundance and deposited bed sediment. Where discharge is constant (for example, in spring fed systems), as macrophytes grow, the decrease in water velocity results in an increase in water depth and the deposition of suspended fine sediment (Collier et al. 1999; Champion and Tanner 2000). The plant roots stabilise this deposited fine sediment, reinforcing the soft-bottomed habitats preferred by macrophytes (Fox 1992). In a New Zealand study of Whakapipi Stream in the Waikato, reach-average water velocity was reduced by 30% in reaches with high submerged macrophyte growth and water depth increased by 40% when compared to reaches where macrophytes were absent (Wilcock et al. 1999). Furthermore, Champion and Tanner's (2000) study in the same catchment estimated average velocity to be reduced by 41%.

Species with poor ability to anchor and those with large hydraulic resistance are most prone to the effects of increased velocity (Franklin et al. 2008). This is particularly an issue at the end of the growing season, where autumn heavy rainfall causes flooding or first frosts initiate senescence and whole plant mats can be uprooted (Janauer 2001). From our observations, this more commonly occurs in *E. guttata* (monkey musk) with bulkier, larger leaves compared to the finer structure of *N. microphyllum* (watercress).

Implications for management

Improving our understanding of the factors that influence macrophyte abundance is helpful in terms of informing alternative management regimes to manage excessive growth in lowland agricultural streams. At the reach scale, the natural disturbance regime is likely the key factor limiting macrophyte growth and physical disturbance and artificial high flow events could be used as alternative macrophyte management tools. At the patch scale, stream shade is a key driver of macrophyte cover, and stream shade is likely the secondary controlling factor after disturbance. Re-establishment of shade by riparian planting, or short-term artificial shading until riparian cover can develop could greatly reduce macrophyte cover. These management approaches have been tested as part of the Canterbury Waterway Rehabilitation Experiment (CAREX; www.carex.org.nz).

Supplement to Chapter 2

Table S2.1. **A,** Reach scale (n = 28) additive linear modelling (lm) of macrophyte cover and reach scale environmental variables to support Figure 2.5. **B,** Patch scale (n = 455) additive linear mixed effects modelling (lmer) of macrophyte cover and patch scale environmental variables including stream reach as a random effect to account for nesting of patches within reaches to support Figure 2.7.

	Estimate	Std. Error	t value	P
A. Reach scale				
lm (Arcsine square-root transformed macrophyte cover ~ nitrate-nitrogen + Dissolved reactive phosphorus + Temperature + pH + Conductivity + Dissolved oxygen + Turbidity + Sediment cover + Wetted width + Discharge)				
(Intercept)	1.59	1.14	1.39	0.18
Nitrate-nitrogen	0.02	0.02	0.83	0.42
Dissolved reactive phosphorus	-0.01	0.01	-1.09	0.29
Temperature	-0.07	0.03	-2.16	< 0.05
pH	0.02	0.15	0.11	0.91
Conductivity	0.00	0.00	0.69	0.50
Dissolved oxygen	-0.01	0.00	-2.58	< 0.05
Turbidity	0.10	0.13	0.80	0.44
Sediment cover	0.00	0.00	2.35	< 0.05
Wetted width	0.08	0.08	1.07	0.30
Discharge	0.18	0.68	0.27	0.79
B. Patch scale				
lmer (Arcsine square-root transformed macrophyte cover ~ Sediment cover + Sediment depth + Stream shade + Bank slope + Water depth + Velocity + Distance to bank + Distance to plants Average substrate size + (1 | Reach), REML = FALSE)				
(Intercept)	0.88	0.23	3.77	< 0.001
Sediment cover	0.00	0.00	2.10	< 0.05
Sediment depth	0.02	0.01	2.81	< 0.01
Stream shade	-0.01	0.00	-7.55	< 0.001
Bank slope	0.00	0.00	0.28	0.78
Water depth	0.00	0.00	0.15	0.88

Velocity	-1.48	0.29	-5.08	< 0.001
Distance to bank	-0.08	0.12	-0.70	0.49
Distance to plants	0.08	0.03	2.64	< 0.001
Average substrate size	0.00	0.00	-0.35	0.73

Plate 3. Time lapse of riparian restoration undertaken at a CAREX site. **Top left,** stream pre-rehabilitation, with overhanging hawthorn (*Crataegus monogyna*) hedge, undercut and collapsing banks, and excessive macrophyte growth. **Top right,** hedge removal. **Centre left,** bank re-battering, fencing setback and community planting day. **Centre right,** landowner meeting. **Bottom left,** two-years post restoration. **Bottom right,** four-years post restoration.

Chapter 3:
Evaluating practical macrophyte control tools on small agricultural waterways in Canterbury, New Zealand

Collins KE, Febria CM, Warburton HJ, Devlin HS, Hogsden KL, Goeller BC, McIntosh AR, Harding JS. 2018. Evaluating practical macrophyte control tools on small agricultural waterways in Canterbury, New Zealand. New Zealand Journal of Marine and Freshwater Research. DOI: 10.1080/00288330.2018.1487454.

Abstract

Excessive macrophytes can cause significant problems in agricultural waterways requiring active management. Conventional control techniques can have a range of adverse effects. We investigated several control tools in two experiments: firstly, we tested eight treatments at a small-scale (2 m x 2 m). We found intensive hand weeding, weed mat and herbicide spraying to be effective treatments, reducing macrophyte cover to < 5 %. Hand weeding and weed mat immediately reduced cover, while dieback from herbicide took two months. Weed mat was a novel and effective control mechanism along stream banks. Secondly, we tested the impact of shading on macrophyte growth. Macrophyte growth was enhanced under partially shaded conditions, but with 80 % effective shading over the entire channel, cover was reduced to 17 %. Once treatments ceased, macrophytes grew back within 3–5 months. Long-term, control methods will require combinations of tools but will need to include optimal shading for the target species.

Keywords: macrophyte control, agricultural drains, agricultural waterways, plant management, aquatic weeds, *Erythranthe guttata; Nasturtium microphyllum*

Introduction

Aquatic macrophytes can provide important ecosystem services, providing habitat for fish and aquatic invertebrates, regulating flow conditions, cycling nutrients and creating sources of carbon (Dawson and Haslam 1983; Sand-Jensen and Mebus 1996; Collier et al. 1999; Fleming and Dibble 2014). However, excessive macrophyte growth can also negatively affect waterways by reducing water flow, increasing sediment deposition, impeding drainage causing flooding, creating large daily fluctuations in dissolved oxygen concentrations resulting in overnight anoxia and altering abundances and community structure of aquatic invertebrates and fish (Fox 1992; Collier et al. 1999; Wilcock et al. 1999; Champion and Tanner 2000; Duggan et al. 2002; Bączyk et al. 2018). Nuisance macrophyte growth is a particular problem in New Zealand agricultural drainage ditches (Hudson and Harding 2004).

Since human settlement in New Zealand, more than 13 million hectares of native forest has been cleared and converted to pastoral agriculture (around 50% of the total land area) (Collier et al. 1995; Quinn 2000). Agricultural land use in Canterbury has greatly intensified since 1990, with large scale conversions from sheep farming and cropping to intensive dairy farming (DairyNZ 2017). Farming is now the most common land use in the mid-lower catchments of many New Zealand rivers (Storey and Cowley 1997; Quinn 2000). This large-scale land-use conversion has resulted in marked changes to waterways. Stream channelisation and wetland drainage has been extensively undertaken in agricultural regions (Collier et al. 1995; Quinn 2000). Many waterways in lowland regions are now modified drains, which are often considered to have little ecological value despite studies showing they can have significant aquatic biodiversity (Armitage et al. 2003; Herzon and Helenius 2008; Sinton 2008; Simon and Travis 2011). These drains often have lowered water quality, including high levels of dissolved nitrogen, phosphorus and sediment and reduced macroinvertebrate and fish species diversity, and importantly can contain nuisance aquatic plants (Greenwood et al. 2012; Burdon et al. 2013; O'Brien et al. 2017).

The aquatic flora of New Zealand streams has changed markedly since human settlement (Reeves et al. 2004). Widespread, rapid deforestation and associated removal of riparian trees and shrubs has resulted in increased light and sediment inputs into streams, allowing the establishment of introduced aquatic macrophyte species (Lacoul and Freedman 2006). Most

New Zealand native macrophyte species form shallow mats and are not vigorous competitors (Reeves et al. 2004). As a result, introduced nuisance macrophytes now dominate many New Zealand lowland streams, including agricultural drains, and native species have either been eliminated or excluded from their preferred habitats (Champion and Tanner 2000; Champion et al. 2002; Reeves et al. 2004).

Historically, drainage has been the primary function of agricultural ditches in lowland New Zealand. Thus, waterways were required to remove excess water efficiently and quickly (Hudson and Harding 2004; Greer et al. 2012). When excessive macrophyte growth occurs, weed management is commonly undertaken (Fox 1992). The three main macrophyte management strategies employed in small flowing waterways in New Zealand are mechanical clearance, chemical sprays and hand weeding. Mechanical clearance typically involves a bankside digger with a scoop bucket physically removing macrophyte biomass (James 2011). This often provides short-term reduction in weed biomass; however, the disturbance resets plant succession and rapid regrowth occurs after clearance, probably resulting in increased establishment (Zehnsdorf et al. 2015). Mechanical clearance is destructive – macroinvertebrates and fish can become entrained and unintentionally removed from the waterway, sediment is released and the practice can spread weeds downstream and between catchments on equipment (Hudson and Harding 2004; James 2011; Greer et al. 2012; Zehnsdorf et al. 2015). Depending on the skill of the operator, it can also cause significant damage to the stream bed and banks (G. Bennett, personal communication). Chemical spraying is less physically intrusive and involves the application of herbicide to kill macrophytes in situ and ranges in scale from bank-based human-operated backpacks through to spraying from a vehicle, boat or helicopter (James 2011). Weed senescence can be significant, though is short lived, and concerns have been raised about the toxicity of residual chemicals and potential de-oxygenation by decomposing plants (Jewell 1971; Brooker and Edwards 1975; Young et al. 2004; James 2011). The third technique, hand weeding, usually involves workers cutting off macrophyte stems using a sickle or scythe and physically removing the plants. This is limited to wadeable waterways. These options all have a level of impact on the waterway, and costs and benefits must be weighed up and the most appropriate methods selected for the specific site and target species (James 2011).

Macrophyte control is an expensive task – in the United States, annual costs were estimated to be $100 million in 2005 (Fleming and Dibble 2014). No cost estimate of macrophyte control is available in New Zealand, although the costs are considerable and expected to be in the tens of millions annually (Hudson and Harding 2004).

We compared a range of tools to control macrophytes and experimentally evaluate their effectiveness at reducing macrophyte cover at a small spatial scale (i.e. metres). Treatments included disturbance to simulate flooding, flower and seed removal (i.e. stopping sexual reproduction), hand weeding, herbicide spray, shading, sediment removal (from the stream bed) and weed mat (Table 3.1, Figure 3.1). We hypothesised that treatments that were more disruptive (e.g. hand weeding and herbicide spray) would be more effective than those which were less disruptive (e.g. weed mat and shading); artificial channel shading to simulate shade provided by riparian planting would limit macrophyte growth by reducing the light available for photosynthesis; sediment removal and artificial disturbance would reduce macrophyte growth by disturbing the habitats that they are able to establish; flower and seed removal would limit macrophyte growth by reducing the ability for seed dispersal; and weed mat would limit growth because the dominant macrophytes in our systems grew primarily from the banks into the water. We also measured the recovery of macrophytes after we ceased treatments to evaluate their ongoing effectiveness.

Materials and methods

Study sites

This study was carried out in two small first order agricultural waterways – Silverstream, near Springston and Boundary Drain, near Ashburton, both in intensive dairying catchments in the Canterbury region, New Zealand. Both streams had spring-fed headwaters, and thus stable flow regimes. Dominant macrophytes in both streams were the introduced emergent species monkey musk (*Erythranthe guttata)* and watercress (*Nasturtium microphyllum*). Other macrophyte species were present in both reaches but at very low biomass and abundance. These two non-native species typically establish roots in the stream bank and extend across the waterway, indicating that control along the banks may provide effective management.

We conducted two experiments, firstly a small-scale macrophyte control experiment was undertaken along a 250-m reach of Silverstream, near Springston (43°39'49.5"S

172°22'27.0"E). Silverstream is fed by springs which emerge from a QEII National Trust covenanted wetland. The springs usually maintain a permanent flow although three consecutive dry years resulted in the springs drying during the second summer of the experiment. Silverstream is a tributary of the Selwyn River which flows into Te Waihora (Lake Ellesmere). The stream reach had an average wetted width of approximately 2 m, depth of 10–20 cm, discharge of 0.01–0.03 m^3/s and cobble substrate covered with a layer of fine sediment. The experimental reach of Silverstream flowed from the NNW to SSE. Prior to our experiment, the reach had a hawthorn (*Crataegus monogyna*) hedge overhanging the stream. This was removed, both banks were rebattered to create gently sloping banks and stop bank collapse, and the riparian zone was fenced to exclude stock and planted with between 2 and 5 m of native plants (including grasses, *Carex* spp., native shrubs and cabbage trees *Cordyline australis*).

The second experiment, a "full shading" trial (i.e. a shade cloth across the entire stream providing 65 – 70 % effective shading), was conducted in a 400-m reach of comparable first-order agricultural waterway on Boundary Drain, Lowcliffe, near Ashburton (44°03'34.4"S 171°36'58.6"E). The stream had a wetted width of 1.5–2 m, a depth of 20–35 cm, a discharge of 0.01–0.04 m^3/s and cobble substrate covered with a layer of fine sediment. The experimental reach of Boundary Drain also flowed from the NNW to SSE. The riparian zone (approx. 2 m) has been fenced to exclude stock; however, the banks lacked extensive riparian planting and were predominantly covered by pasture grass and exotic weed species, and a few isolated *Carex* spp.

Study design and field methods

Small-scale macrophyte control experiment

This experiment compared the effectiveness of multiple macrophyte control methods. To do this we tested eight treatments each with seven replicates by establishing fifty-six 2 m x 2 m plots along a 250-m reach of upper Silverstream (Table 3.1, Figure 3.1). The plots were blocked and then randomly assigned to each of the eight treatment groups. One metre of each plot was on the stream bank, and the second half extended into the water for 1 m. At least 2 m separated adjacent plots. Plots were located to minimise the effects of the riparian planting on the treatments.

Table 3.1. Experimental treatments trialed in the small-scale macrophyte control experiment.

Treatment	**Description**	**Frequency**
Control	No treatment	NA
Disturbance	The plot was raked over vigorously to disturb fragments and simulate a flood	Monthly
Flower removal	All macrophyte flowers and seed heads within the plot removed by hand to stop sexual reproduction and limit recruitment	Monthly
Hand weeding	All visible plant and root materials were removed using garden trowel	Monthly
Herbicide spray	Glyphosate 510 (AGPRO) was sprayed directly onto emergent macrophytes at the manufacturers' recommended rate	Every two months or as weather permitted
Partial shade	A steel post was installed at either end of the plot at a 45° angle. Wires were run between the posts and a piece of 1.83 m wide 65–70 % medium shade level knitted shade cloth (Egmont Commercial) was attached to evaluate the effect of shading on macrophyte growth	Installed in October 2014
Sediment removal	Fine sediment (<1 mm) and any macrophytes in the wetted section of the plot were disturbed through kicking to reduce cover to less than 20 %, to test if fine sediment in the bed enhanced macrophyte growth	Monthly
Weed mat	Weed mat was installed over entire dry area of plot and extended into the waterway. Initially, EcoWool mulch matting 500 gsm weed mat (Advance Landscape Systems) was used, pegged down with plastic pegs. This weed mat broke down quickly, and was replaced in January 2015 with more EcoWool. In April 2015, it was replaced again with plastic woven 100 gsm weedmat (Egmont Commercial) because it was a more hard-wearing and long life product	Installed in October 2014, replaced after 3 and 6 months

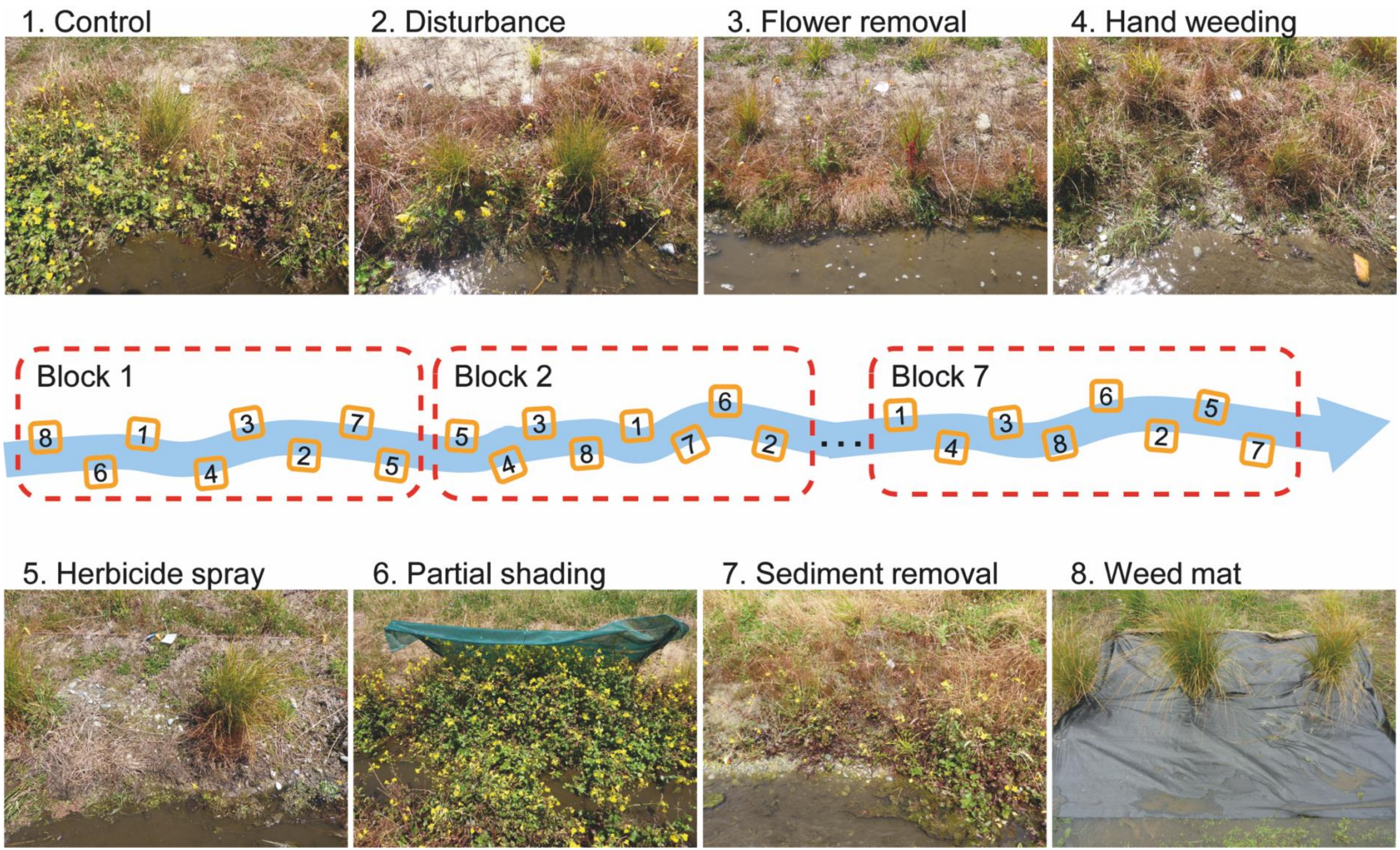


Figure 3.1. Study design of small-scale macrophyte control experiment, showing photos of the eight treatments and the random blocked design of the experiment with seven blocks, each containing one replicate of each treatment.

Prior to treatment in October 2014, macrophyte percentage cover was measured by dividing each 4 m^2 plot into 16 quadrats. In each quadrat macrophyte cover was estimated visually. Because the stream banks had recently been rebattered (i.e. bank slopes had been altered to reduce erosion), macrophyte cover was between 10% and 20% at the beginning of the experiment. Macrophyte percentage cover was measured monthly from October 2014 to July 2015 (growing season one), but in July 2015, the disturbance, partial shade and sediment treatments ceased. The remaining five treatments were continued for a further 15 months (i.e., a second growing season, 24 months total), then in October 2016, the weed mat was removed, and the hand weeding, flower and seed removal and herbicide spray ceased. Plots associated with the remaining five treatments were monitored for a further five months over a summer growing season to measure the recovery of macrophyte growth.

To investigate the light reduction provided in the partial shade treatment, pairs of light loggers were set to record temperature (°C) and light intensity (lumens/ft^2) every 30 min (HOBO onset pendant temp/light logger). Long et al. (2012) found comparable estimates of light intensity recorded from HOBO loggers when compared to PAR sensors. Loggers were installed at two randomly selected shade plots, one on each side of the stream. One logger from each pair was placed under the centre of the bankside top edge of the partial shading plot, which was the most shaded portion of the plot, and the second was placed 2 m upstream of the plot.

Full shading trial

This trial simulated riparian shading across the entire stream channel by using 65 – 70 % medium shade cloth to evaluate how effective shade is at controlling macrophytes. To do this, three 5-m shade tunnels, and three unshaded 5-m control reaches were set up in December 2014 in a single stream reach of Boundary Drain in South Canterbury; control reaches were upstream of the shaded reaches. Five metre lengths of 65–70% knitted shade cloth, 3.66 m wide (source Egmont Commercial Limited) were placed over the waterway; 4 x 4.1 m long, 10 mm diameter, fibreglass rods (Polynet Products Limited) were hooped over the waterway (at right angles to water flow) every 1.25 m, as a frame to support the shade cloth (Figure 3.2).

Within each shade tunnel and control reach, three permanent macrophyte assessment transects were set up 1 m apart. Prior to shade cloth installation, macrophyte transects were measured in

December 2014, and then monthly for 7 months. When the experiment started, macrophytes were already well established and had between 75% and 100% cover and extending 30–50 cm above the water surface. Shade tunnels were removed in July 2015 and transects were measured for a further 8 months to measure the recovery of macrophytes.

In each assessment transect, macrophyte species were identified and their height above the water surface was measured with a ruler at every 10 cm across the transect, thus each transect had about 15 – 20 measurements.

To investigate the light reduction provided by the shade tunnels, a pair of light loggers were set to record temperature (˚C) and light intensity (lumens/ft^2) every 30 min (HOBO onset pendant temp/light logger). One logger was placed under the shade tunnel, in the centre pegged to the stream bank, and the second was attached to an adjacent fence post.

Statistical analyses

In the small scale experiment, changes in macrophyte cover between treatments over time were assessed using repeated measures analysis of variance (ANOVA) testing using a block term, date and treatment as factors and a treatment x date interaction. A block by plot error term was used to ensure the correct residual error term for repeated measures ANOVA. One way ANOVA testing including a blocking term was undertaken on single dates during the trial. Post hoc comparisons of means were made using Tukey HSD tests.

In the full shading trial, changes in macrophyte cover and height between control and shade treatments over time were assessed using repeated measures ANOVA testing using date and treatment as factors and treatment reach as an error term. One-way ANOVA testing was undertaken on the date shading was removed.

All statistical analyses were undertaken using R statistical software version 3.1.2 (R Core Team 2014) and macrophyte percent cover values were normalised by arcsine square-root transformation (Zar 2009).

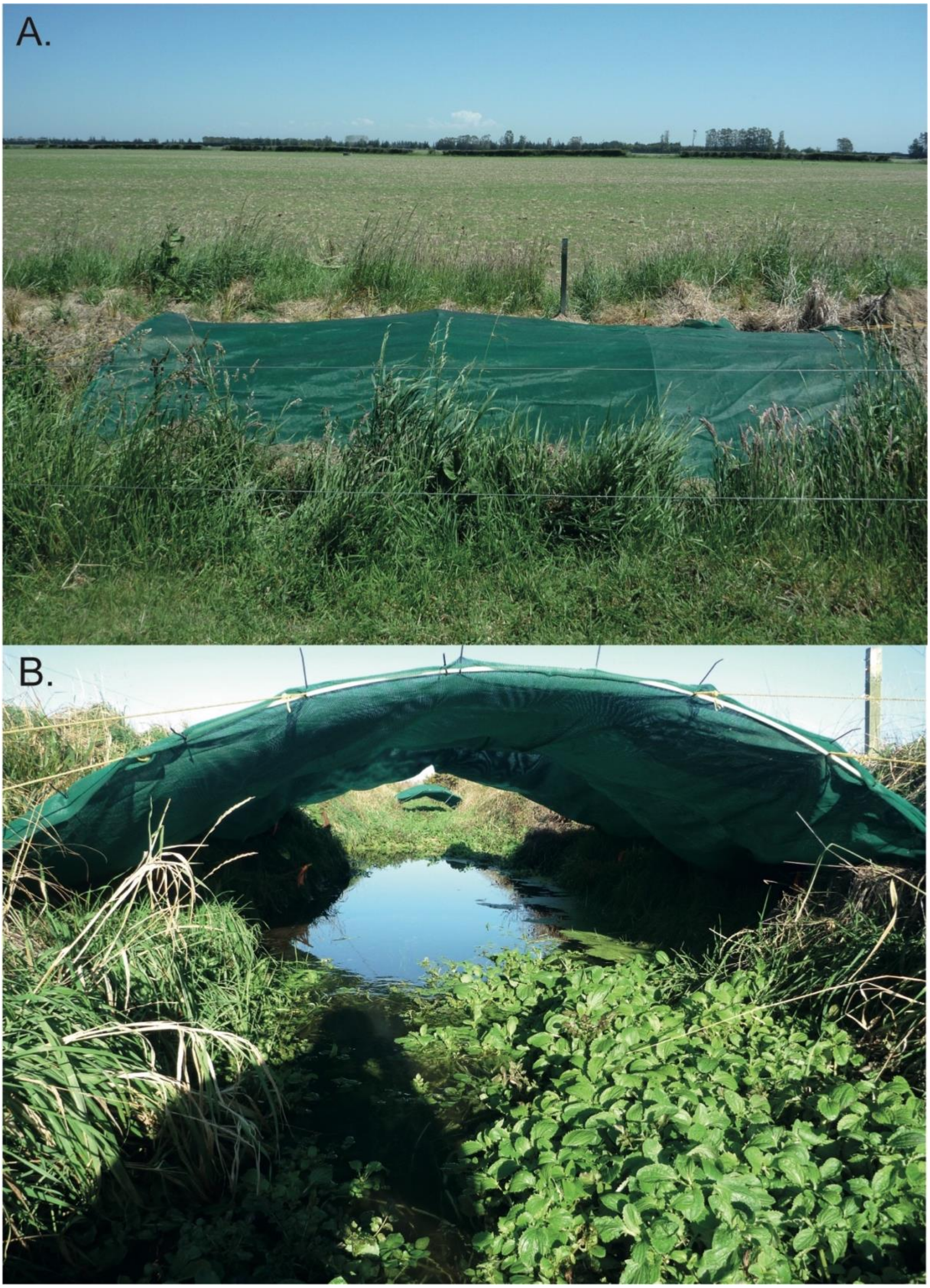


Figure 3.2. Experimental setup of full shading trial. **A**, Shade tunnel setup showing shade cloth installed over fiberglass rods on December 2014. **B**, Effectiveness of full shade treatment after 6-months in June 2015.

Results

Small-scale macrophyte control experiment

The results of the small-scale macrophyte control experiment showed statistically significant differences among the eight treatments over time (significant treatment * time interaction, $F_{63, 432} = 15.94$, $P < 0.001$) over the first growing season (summer 2014–15) (Figure 3.3A, Table S3.1). The untreated control plots showed a predictable seasonal growth pattern, with macrophyte cover peaking in summer and autumn ($\bar{x}$ 37% ± 6% 1 SE), and then dying back in winter (Figure 3.3A, Table S3.1). Herbicide, weed mat and hand weeding were very effective treatments, showing a marked reduction in macrophyte cover to <5% (Figure 3.3A). The hand weeding and weed mat treatments resulted in immediate reductions in macrophyte cover, while dieback from herbicide spray did not occur until two months after spraying. The macrophyte disturbance and sediment removal treatments also reduced macrophyte cover relative to the untreated control but were not as effective as the previous three treatments and resulted in cover between 10% and 20% across all seasons (Figure 3.3A). In contrast, the flower and seed removal treatment had no effect on macrophyte cover and showed a similar seasonal pattern to the untreated control. Surprisingly, the partial shading treatment resulted in enhanced macrophyte growth relative to the control, reaching a peak of 50% cover (Figure 3.3A).

At the peak of macrophyte growth in autumn, there was a statistically significant effect of treatment on macrophyte cover ($F_{7, 42} = 56.31$, $P < 0.001$). Post hoc Tukey tests showed that compared to the control there was no difference in macrophyte cover in the flower removal or partial shading treatments. The sediment removal and disturbance treatments showed a decrease in macrophyte cover from the control. The greatest decrease in macrophyte cover were seen in the herbicide spray, hand weeding and weed mat treatments (Figure 3.3B, Table S3.1).

During the second growing season (summer 2015–16), a similar seasonal pattern was observed in macrophyte cover in response to treatments (significant treatment * time interaction, $F_{116, 870} = 9.50$, $P < 0.001$) (Figure 3.4, Table S3.2). Macrophyte cover was higher in control plots over summer 2015–16 (ranging from $\bar{x}$ 48%–67%) compared to summer 2014–15 ($\bar{x}$ 31%–37%). Peaks in macrophyte cover in the herbicide treatment occurred in autumn 2016 and to a lesser extent in summer 2015 – 16, whereas macrophyte cover in the weed mat treatment peaked in summer 2015–16 and winter 2016. These peaks coincided with both periods of poor weather which made spray application unsuitable and time when the weed mat broke down. Growth in hand weeding plots remained low at <10% across all seasons.

In spring 2016, all macrophyte suppression treatments were stopped and macrophytes rapidly recovered in the herbicide spray, weed mat and hand weeding treatments (Figure 3.4A). At the cessation of treatments, there was a statistically significant effect of treatment on macrophyte cover ($F_{4, 24} = 70.29$, $P < 0.001$) (Table S3.2). Post hoc Tukey tests show that macrophyte cover in the control and flower removal treatments were statistically the same, while cover in the herbicide spray, hand weeding and weed mat were also the same but significantly reduced (Figure 3.4B). However, macrophyte recovery in the weed mat treatment occurred at a faster rate than recovery in the herbicide and hand weeding treatments (Figure 3.4A). On the final date of measurement, there was still a statistically significant effect of suppression treatment on macrophyte cover ($F_{4, 24} = 3.97$, $P < 0.05$) (Table S3.2); however, post hoc Tukey tests showed that the only remaining significant difference was that cover in weed mat plots had statistically more macrophyte cover than that in hand weeding (Figure 3.4C).

Differences in light intensity were observed between plots on different sides of the stream (Figure 3.5). On the true right bank, the light intensity peaked earlier in the day and reached a higher level compared to that measured on the true left bank. In the evening on the true left bank, light levels were elevated for a further hour compared to those on the right bank. Light loggers showed there was up to 65 % light reduction under the most shaded portion of the shade cloth. Under shade on the true right bank, there was a peak in light intensity in the morning, then a drop off before another peak and then drop off into evening (Figure 3.5). The reverse pattern, with a peak in the evening, was seen on the true left bank.

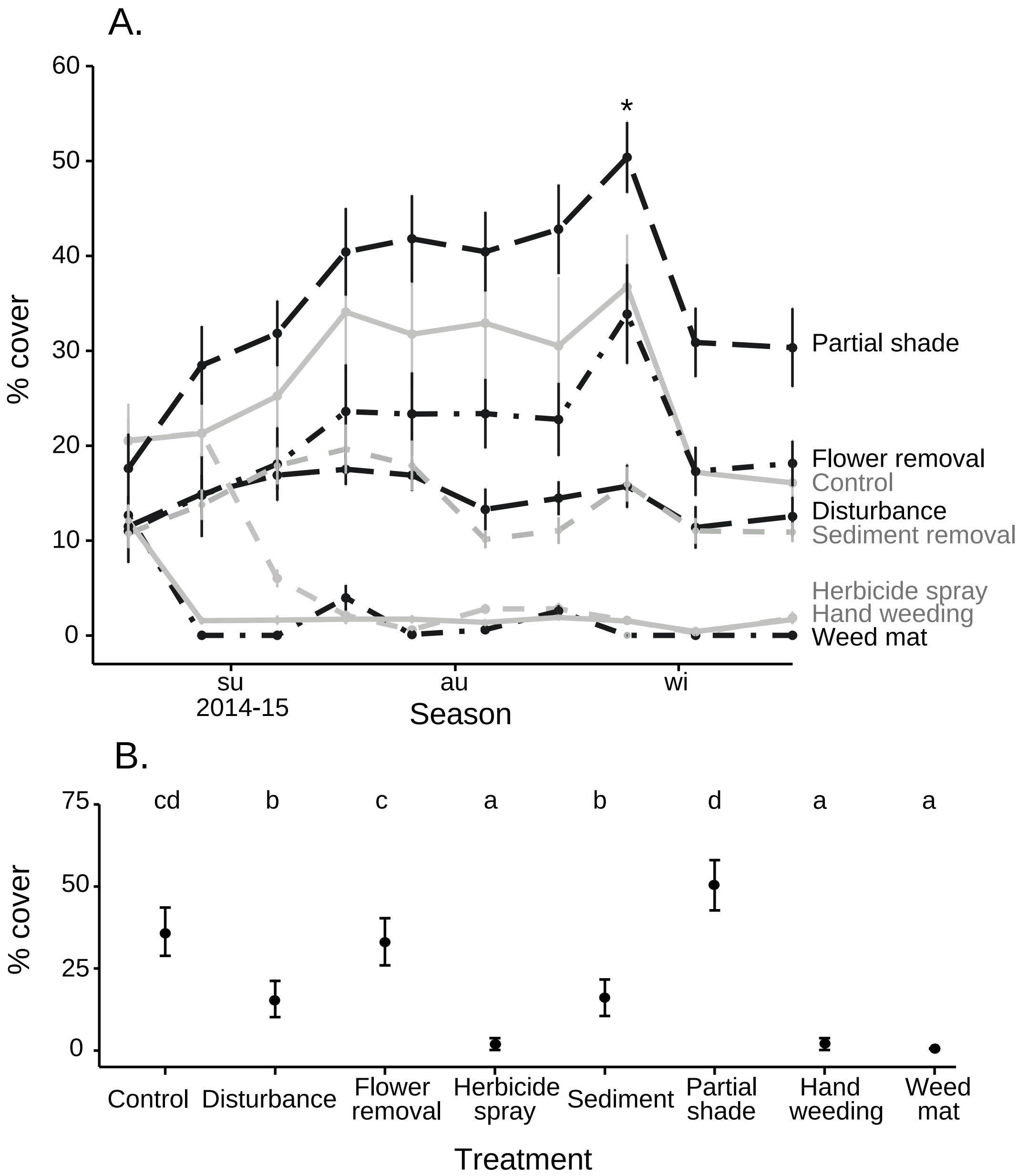


Figure 3.3. Macrophyte cover over eight treatments testing various macrophyte control treatments (n = 7). **A**, Mean (± SEM) macrophyte cover for the 2014–15 growing season. **B,** Post hoc Tukey tests at the height of macrophyte growth (date of testing indicated by * in A). Letter values above $\bar{x}$ ± 95% CI error bar in B indicate statistically significant differences between individual treatments at peak macrophyte cover.

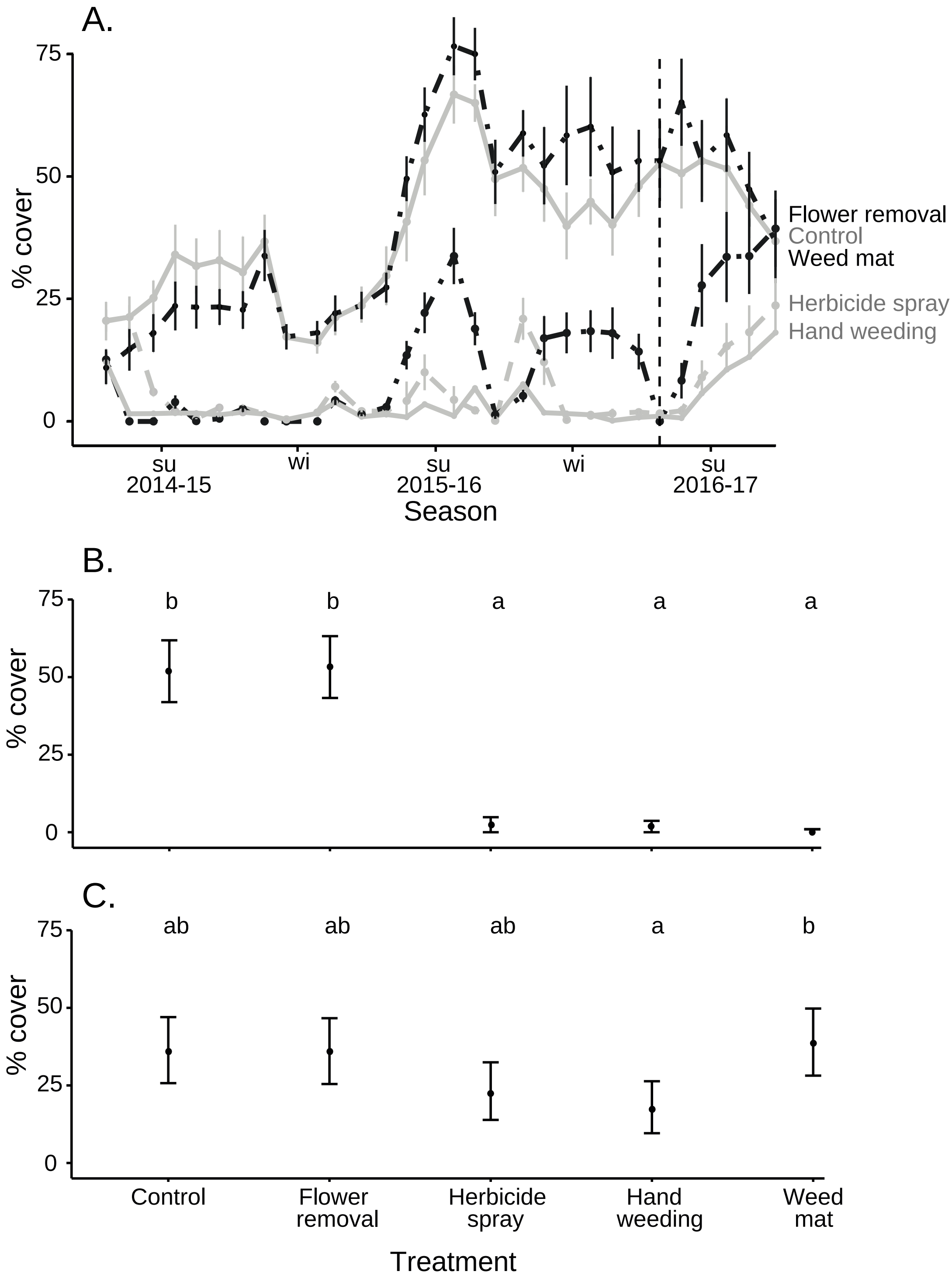


Figure 3.4. Macrophyte cover over five treatments testing various macrophyte control techniques (*n* = 7). **A**, Mean (± SEM) macrophyte cover from October 2014 – March 2017. **B,** Post hoc Tukey tests on the date treatment ceased (indicated by dashed vertical line in [A]). **C,** Post hoc Tukey test on the final date. In A, dashed vertical line indicates when treatments ceased in October 2016 to allow for macrophyte recovery post treatment to be measured. In B and C, letter values above $\bar{x}$ ± 95% CI error bar indicate statistically significant differences.

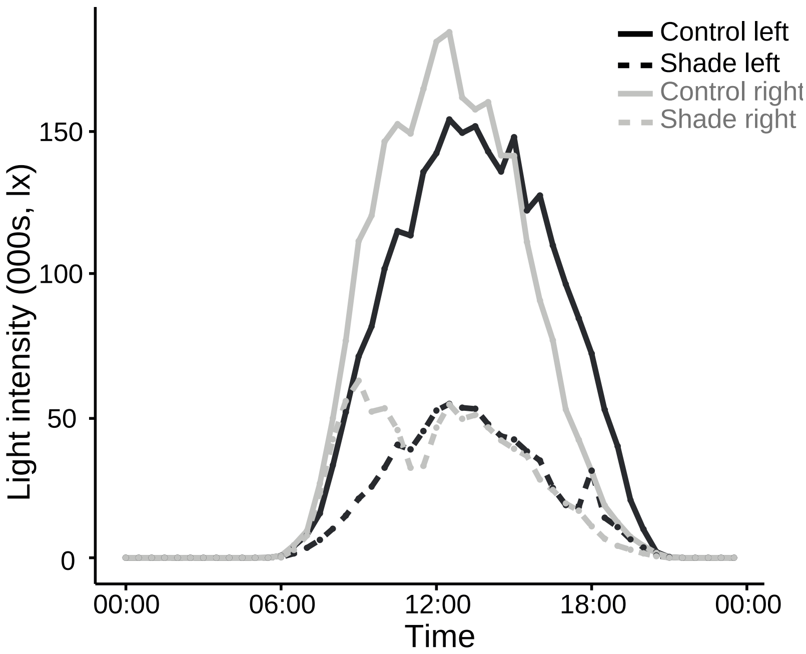


Figure 3.5. Diel fluctuation in light intensity between control and partially shaded plots on both right and left stream banks. Lines are daily averages of light intensity data collected from 10 November – 5 December 2014.

Full shade trial

Initially, mean macrophyte cover and height in all reaches were high, covering around 90% and 40 cm above the water surface at the beginning of the full shade trial (Figure 3.6). The effect of shading on macrophyte cover (significant treatment * time interaction, $F_{15, 60} = 5.05$, $P < 0.001$) and macrophyte height (significant treatment * time interaction, $F_{15, 60} = 3.91$, $P < 0.001$) changed over the experiment, with both cover and height being reduced markedly by shading (Table S3.3).

Eight months after shade tunnels were installed, macrophyte cover was reduced to $\bar{x}$ 17% (7% ± 1 SE) (Figure 3.6A). In contrast, cover was $\bar{x}$ 65% (12% ± 1 SE) in the unshaded controls. At the beginning of the trial macrophyte cover did not respond to shading for two months; however, between February and March, macrophyte cover declined under the shade treatment by 40%, and by a further 30% between March and May. Maximum die off was reached after five months, when cover stabilised between 20% and 30% until the trial ended and shading was removed. In contrast, reductions in macrophyte height started to occur in the month following the installation of shading (Figure 3.6B). Three months post shade installation, macrophyte height was reduced to below 10 cm, then to 0–3 cm, where it was maintained until shading was removed. In the unshaded control reaches, while macrophyte cover remained in excess of 60% over winter, macrophyte height was greatly reduced to less than 10 cm above the water surface during that time.

We also compared the response of the two dominant macrophyte species: monkey musk and watercress. Monkey musk cover took two months to be reduced by shading (Figure 3.6C). However, watercress showed a decline in cover immediately after shading (Figure 3.6D). Three months post shading, watercress was eliminated from the shaded reaches, and monkey musk cover remained below 15% for the remainder of the trial.

At the start of spring 2016, the shade tunnels were removed from the shaded reaches. On this date, there was a statistically significant effect of shading on macrophyte cover ($F_{1, 4} = 11.5$, $P < 0.01$) (Table S3.4). However, the height of macrophytes was not significantly different to the control, and nor was the cover of monkey musk or watercress. Macrophyte cover and height were suppressed and cover was maintained between 20% and 30% cover for three months post shade removal (Figure 3.6). Between three and seven months after shade removal monkey musk recovered rapidly. Noticeably, when shade was removed watercress did not recover to control levels; however, watercress in the control reaches was also low during this period.

Light loggers showed there was up to 80 % light reduction under the shade tunnels (Figure 3.7).

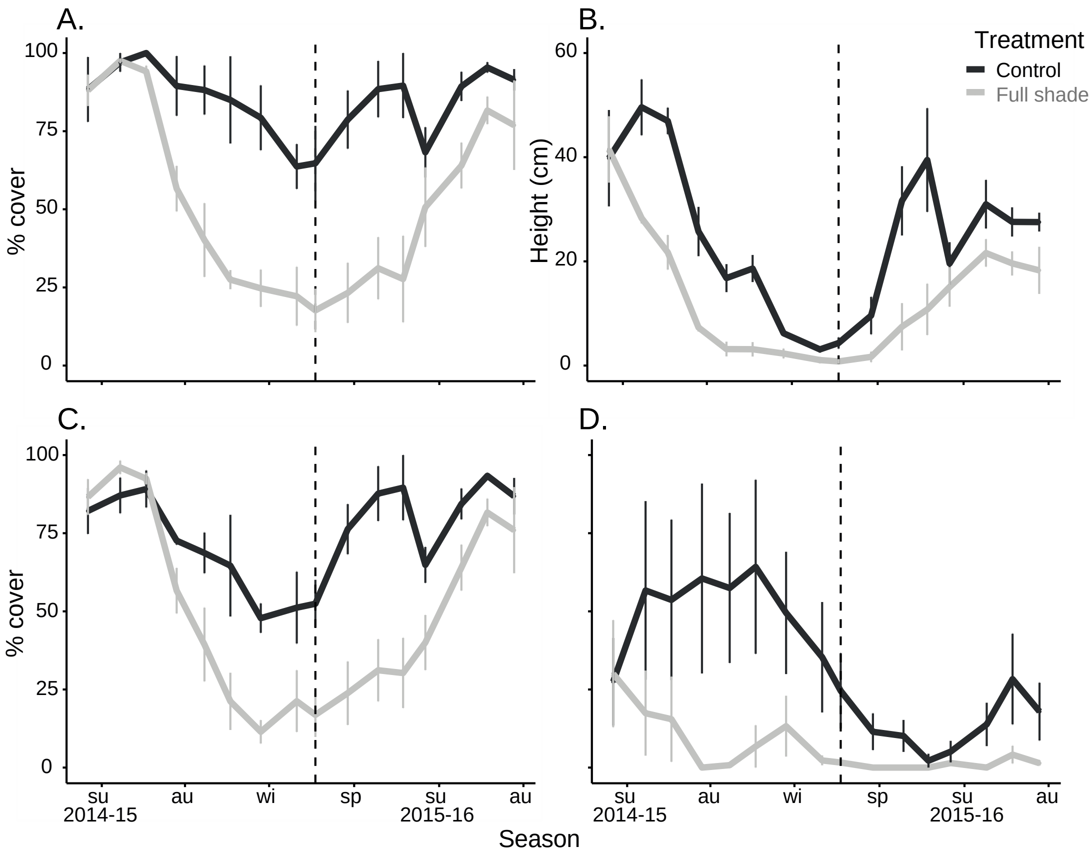


Figure 3.6. Macrophyte cover and height between control and full shade reaches from December 2014 – March 2016 (n = 3). Shade was removed in July 2015 (dashed vertical lines) and reaches were continued to be monitored to establish the rate of macrophyte recovery post shading. **A,** Mean (± 1 SEM) macrophyte cover. **B,** Mean (± 1 SEM) macrophyte height. **C,** Mean (± 1 SEM) monkey musk cover. **D,** Mean (± 1 SEM) watercress cover.

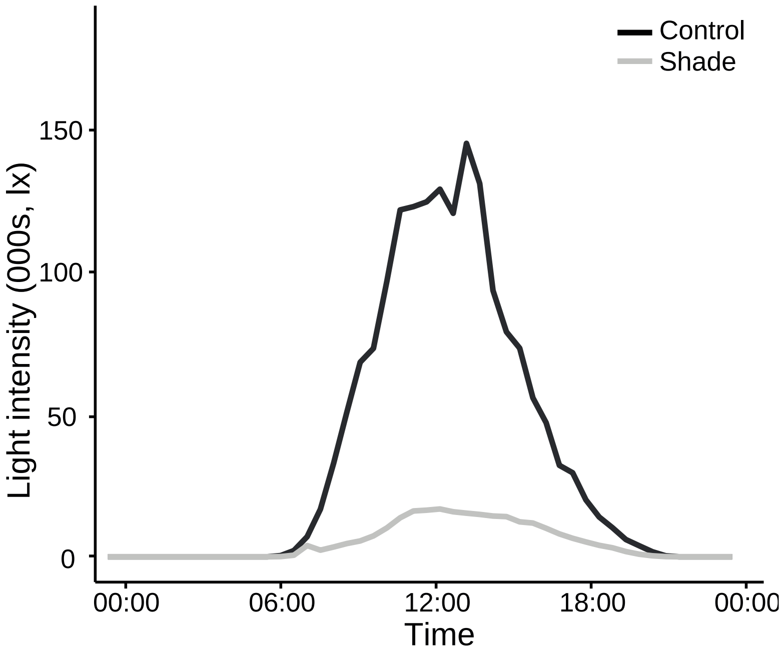


Figure 3.7. Diel fluctuation in light intensity between control and full shade tunnels. Lines are daily averages of light intensity data collected from 22 January – 18 February 2015.

Discussion

Overall, we found mixed effectiveness of the macrophyte control techniques we tested. Intensive hand weeding, weed mat and herbicide spray provided the most effective macrophyte control at a small scale, followed by shading across the entire stream channel providing 80 % effective light reduction. We discuss these control methods and their associated management implications below.

Of the seven management approaches we trialled, herbicide spray (i.e. glyphosate) was found to be an effective control. Glyphosate is one of the world's most effective and most frequently used herbicides. It is absorbed through leaves and is transported by the plant to the growing points in roots and shoots, where it prevents the plant from being able to synthesise proteins that are required for growth (Magbanua et al. 2013). It is a non-selective, broad-spectrum herbicide commonly used on emergent marginal macrophytes (Solomon and Thompson 2003). Following manufacturer's instructions, spraying directly on the waterway should be minimised due to the potential adverse toxicity effects on aquatic organisms (Hudson and Harding 2004); however, investigations undertaken by Folmar et al. (1979) suggested glyphosate application at recommended rates on emergent marginal vegetation should not affect fish or macroinvertebrates. Our experiment showed that glyphosate is an effective treatment in reducing weed cover for two to three months. However, regular spraying was needed to suppress our two common macrophyte species: monkey musk and watercress. We also recorded a lag effect of two months between the application of spray and observed macrophyte senescence. From this study we suggest the timing of spraying within the growing season is important, where targeting macrophyte seedlings early in their growth stages reduces cover before rapid growth occurs. Spraying must be undertaken on a fine and calm day with little wind and dry weather for at least the following 24 hours, which can limit appropriate days for spray application and allow macrophyte growth to take off. The effects of these constraints can be seen in the peaks in macrophyte cover in our herbicide treatment which coincided with periods where spraying was not able to be undertaken regularly enough to ensure control. Therefore, although herbicide application reduced macrophyte cover, its potential adverse toxicity effects (as noted on the packaging by the manufacturer), issues surrounding the effects of spray drift into areas of organic farming and application constraints may limit its overall suitability for broad-scale macrophyte control in agricultural regions.

Our weed mat treatment was also effective at controlling bank-based sprawling emergent macrophytes over two growing seasons. However, maintenance of the weed mat was an issue. Macrophyte colonisation occurred from the edges of the weed mat, and sedimentation on top of the weed mat also provided substrate that allowed reestablishment. However, in our study we used small plots which were liable to be affected by edge effects. Weed mat or benthic barriers created using both plastic sheeting and biodegradable jute matting have been successful in controlling submerged aquatic macrophytes in lakes in both New Zealand and overseas (Ussery et al. 1997). However, we found no published information of the application of weed mat to banks of flowing waters to control emergent and marginal macrophytes. In lakes, plastic matting is buoyant and can affect macroinvertebrate communities, reduce oxygen levels and restrict nutrient exchange (Ussery et al. 1997; Caffrey et al. 2010; Hofstra and Clayton 2012); thus we initially used a biodegradable wool matting. Biodegradable matting allows water and gas exchange, and the natural breakdown is ecologically preferable to plastic non-degradable matting (Barr and Ditomaso 2014). Caffery et al. (2010) found biodegradable jute matting effective in eradicating the invasive macrophyte *Lagrosiphon major* from an Irish lake while allowing native macrophytes to reestablish through the weave in the matting. They found that the matting retained its integrity 10 months after placement (Caffrey et al. 2010); however, we found the EcoWool product that we used broke down within three months, and eventually had to be replaced with woven plastic weedmat, which did not break down. Our results indicate that biodegradable wool weed mat has a very limited life span on stream banks; however, woven plastic weed mat can provide continued macrophyte control.

Like weedmat, hand weeding was effective at controlling macrophytes, however this required regular, ongoing maintenance because 1-2 seedings were removed from each treated plot every month. Hand weeding is one of the most targeted macrophyte control options, where nuisance plants can be removed while desirable plants are left intact. Hand weeding has been successfully used to eradicate small infestations of unwanted plant species; however, it is a very labour and cost intensive technique (Hussner et al. 2017). Given that hand weeding provided similar short-term reductions in macrophyte coverage to herbicide spray and weed mat but had to be repeated monthly to produce sustained reductions in coverage. Consequently, the application of repeated hand weeding should be limited to small areas of particularly invasive pest plants and should not be used for broad-scale macrophyte control.

Both the disturbance and sediment removal treatments reduced macrophyte growth relative to the control. However, they do not provide total macrophyte control because macrophyte cover persisted between 10% and 15%. These control methods altered the fitness of the macrophytes by disturbing them and altering the habitat. Macrophyte cover and species richness have been found to decrease at sites where floods were more frequent (Riis and Biggs 2003; Bowden et al. 2007). Riis and Biggs (2003) showed that macrophytes did not occur at sites with more than 13 annual events where flow was seven times greater than the median. Our disturbance treatment damaged plants in a similar way to that which might be sustained through flooding, although the waterway we tested this in was spring fed, with stable, seasonal base flows. Our sediment treatment removed excessive sediment from the stream bed to reduce the ability of macrophytes to establish. Macrophytes in streams are typically associated with sediment, which is a favoured substrate to root (Fox 1992). Furthermore, a feedback loop probably occurs where macrophyte stems and roots trap entrained sediment and increase deposition and accumulation of sediment, reinforcing the soft-bottomed habitats they prefer (Wood and Armitage 1997; Jones et al. 2012). In the absence of natural disturbances by floods to scour macrophytes and sediment, management tools to increase stream bed disturbance may help control macrophytes.

We did not find flower and seed removal to be an effective macrophyte control treatment. Over the first growing season, any reduction in cover was not expected; however, reductions in macrophyte cover were also not seen over subsequent years as a response to reduced seed availability. While a seasonal pattern of growth was seen over the experiment, macrophyte cover remained over 20 % in winter and plants did not completely die off. Monkey musk and watercress produce large numbers of small seeds (monkey musk seeds are c. 0.02 mg, 0.5 mm wide x 1 mm long with an average of 7000 seeds released per stem) that are effectively distributed downstream by flow, and generally by birds and wind (Vickery Jr et al. 1986; Truscott et al. 2006). Our flower removal treatment removed flower and seed heads, eliminating the ability for the plants to reproduce sexually. In environments where macrophytes completely die off over winter, they rely on sexual reproduction to regenerate. However, in wet environments, both species are able to grow from stolons and vegetative fragments in addition to seed production (Waser et al. 1982; Dole 1992; Truscott et al. 2006). This ability to grow from fragments likely enabled recovery in macrophyte cover in the absence of direct seed availability in our experiment. Furthermore, areas adjacent to treated plots were producing seeds that may have dispersed into treated plots, and plants may have expanded in from adjacent areas or fragments deposited from upstream by flows. Overall, because of seed dispersal from plants outside the experimental area, as well as the lack of complete macrophyte die off in winter, flower and seed removal was of little use to control macrophyte growth.

Partial shading provided by bankside vegetation has been suggested as the optimum for allowing macrophyte presence for ecosystem function while preventing excessive growth (Dawson and Haslam 1983). However, we found that partial shading created a microclimate that supported and enhanced macrophyte growth, allowing in enough light to enable growth, while providing protection from excessive sun and wind. This shows the importance of scale and placement in implementing shading. In comparison, our shading trial providing 80 % effective shading across the full channel showed that macrophyte cover and height are both reduced when the stream channel is fully shaded with 80 % light reduction. Where streams are fully shaded, there are few or no macrophytes present because only a few species are adapted to grow in forested streams (Champion and Tanner 2000; Bowden et al. 2007). However, due to large-scale land clearance in New Zealand, there are very few lowland streams shaded by tall plants. This is especially true with the rise in centre-pivot irrigation systems in much of the South Island, where plant growth must be kept below rotating sprinklers at 2–3 m in height. Therefore, riparian shading controls must be strategically implemented to provide enough riparian shade to suppress but not enhance macrophyte growth.

Our shade-related findings indicate that light strongly affects macrophyte growth (Dawson and Haslam 1983) and is arguably the predominant factor that limits their distribution and abundance (Bunn et al. 1998). Macrophyte growth can be greatly reduced in streams where shade is reinstated through riparian management (Dawson and Haslam 1983); however, complete eradication is challenging (Hussner et al. 2017). The intensity of shading required to control macrophytes is not well understood and is species-specific (Hussner et al. 2017). de Winton et al (2013) suggest that physical shading needs to filter out 90% available light for macrophyte control to be effective. However, we have shown that 80 % shading is sufficient to reduce monkey musk cover to < 20 % and remove watercress altogether in our waterways. This level of cover allows for an ideal balance, providing for beneficial macrophyte functions including habitat for fish and invertebrates, regulating flow conditions, cycling nutrients and creating carbon sources (Dawson and Haslam 1983; Sand-Jensen and Mebus 1996; Collier et al. 1999; Fleming and Dibble 2014) without reaching problem levels that reduce flow, encourage sediment deposition, impede drainage and cause large daily dissolved oxygen fluctuations (Fox 1992; Collier et al. 1999; Wilcock et al. 1999; Champion and Tanner 2000; Duggan et al. 2002; Bączyk et al. 2018). These results indicate that optimum macrophyte control by shading may be obtained without complete restoration of full riparian cover, which may be impractical in some cases.

In our full shade experiment, we saw a lag effect of two months before macrophyte cover began to decline; however, macrophyte height was reduced in the first month. These patterns are similar to those reported by Dawson and Hallows (1983), where time to complete macrophyte control ranged from 5 to 12 weeks depending on species, and morphological changes were observed during the period to control including smaller leaves and stunted growth (Dawson and Hallows 1983). Monkey musk percent cover under full shading shows a similar pattern to the combined cover because it is the dominant macrophyte species in this stream and is driving the overall pattern. In contrast, watercress percent cover declined immediately after shading. Hence, the response of macrophytes to shading may be species-dependent, with different morphological responses and lag times associated with shading.

Once we stopped all treatments, macrophytes started to grow back regardless of the treatment. In our small-scale trial, regrowth in the herbicide, weed mat and hand weeding treatments occurred within a few months. This was likely due to the residual seed bank, the large number of small, mobile seeds produced annually, invasion from adjacent areas and establishment of plant fragments that have floated from upstream. In the full shading trial, as soon as we removed the shade, macrophyte growth increased rapidly, recovering to 75 % macrophyte cover within seven months. However, watercress did not recover after shade removal. This rapid regrowth (within one growing season) highlights the importance of regularly undertaking macrophyte management in the absence of permanent shading and applying the right management intervention at the right time.

Conclusions

We were able to confirm several effective techniques for small-scale macrophyte control. Intensive hand weeding, weed mat and herbicide spraying were found to be effective treatments, reducing macrophyte cover to < 5 %. Although weed mat has been deployed in lakes to control macrophytes, we do not believe this has been applied along the banks of flowing waters. We have demonstrated that this is a novel and effective macrophyte control mechanism along stream banks. Interestingly, while hand weeding and weed mat resulted in an immediate reduction to macrophyte cover, the dieback from herbicide spray took two months to occur. Shading over the full channel providing 80 % effective light reduction reduced macrophyte cover by 50 % to 17 % cover. This level of macrophyte cover allows for an ideal balance, providing for the beneficial functions that macrophytes provide in stream systems without reaching problem levels that impede drainage and cause flooding.

Longer term control is difficult when recolonisation by macrophytes can be achieved by seed dispersal, seed banks in the riparian zone and spread by fragments. This explains why common mechanical clearance techniques do not provide longer term control. Physical disturbance through flooding and sediment removal does reduce growth, and high flows could be used as a management tool to reduce macrophytes. Interim techniques to provide immediate control, incorporating weed mat, herbicide spray or hand weeding may be required until sufficient shading can be achieved through riparian planting to control macrophyte growth.

Supplement to Chapter 3

Table S3.1. ANOVA output tables for macrophyte cover over eight treatments for the 2014–15 growing season to support Figure 3.3.

	d.f.	SS	MS	F	P
A. Repeated measures ANOVA of macrophyte cover for the 2014-15 growing season					
aov(Macrophyte cover ~ Treatment * Date + Block + Error (Block_Plot)					
Error: between					
Treatment	7	18.63	2.41	47.56	< 0.001
Block	6	0.41	0.07	1.21	0.32
Residuals	42	2.35	0.06		
Error: within					
Date	9	0.74	0.08	21.37	< 0.001
Treatment x date	63	3.87	0.06	15.94	< 0.001
Residuals	432	1.67	0.004		
B. One-way ANOVA at the height of macrophyte growth					
aov(Macrophyte cover ~ Treatment + Block)					
Treatment	7	4.00	0.57	56.31	< 0.001
Block	6	0.02	0.003	0.33	0.92
Residuals	42	0.43	0.01		

Table S3.2. ANOVA output tables for macrophyte cover over five treatments from 2014–17 to support Figure 3.4.

	d.f.	SS	MS	F	P
A. Repeated measures ANOVA of macrophyte cover from 2014 – 2017					
aov(Macrophyte cover ~ Treatment * Date + Block + Error (Block_Plot)					
Error: between					
Treatment	4	56.96	14.24	94.66	< 0.001
Block	6	3.26	0.54	3.61	< 0.01
Residuals	24	3.61	0.15		
Error: within					
Date	29	14.96	0.52	37.58	< 0.001
Treatment x date	116	15.13	0.13	9.50	< 0.001
Residuals	870	11.94	0.01		
B. One-way ANOVA on the date treatment ceased					
aov(Macrophyte cover ~ Treatment + Block)					
Treatment	4	4.65	1.16	70.29	< 0.001
Block	6	0.26	0.04	2.61	< 0.01
Residuals	24	0.40	0.02		
C. One-way ANOVA on the final date of measurement					
aov(Macrophyte cover ~ Treatment + Block)					
Treatment	4	0.33	0.08	3.97	< 0.01
Block	6	0.68	0.11	5.49	< 0.001
Residuals	24	0.49	0.21		

Table S3.3. ANOVA output tables for macrophyte cover and height between control and full shade reaches from 2014–16 to support Figure 3.6.

	d.f.	SS	MS	F	P
A. Repeated measures ANOVA of macrophyte cover between control and full shade reaches from 2014 – 2016					
aov(Macrophyte cover ~ Treatment * Date + Error (Site)					
Error: between					
Treatment	1	4.54	4.54	12.16	<0.05
Residuals	4	1.49	0.37		
Error: within					
Date	15	5.23	0.35	17.34	<0.001
Treatment x date	15	1.52	0.10	5.05	<0.001
Residuals	60	1.21	0.02		
B. Repeated measures ANOVA of macrophyte height between control and full shade reaches from 2014 – 2016					
aov(Macrophyte height ~ Treatment * Date + Error (Site)					
Error: between					
Treatment	1	3502	3502	17.87	<0.01
Residuals	4	784	196		
Error: within					
Date	15	14008	933.8	29.10	<0.001
Treatment x date	15	1881	125.4	3.91	<0.001
Residuals	60	1925	32.1		
C. Repeated measures ANOVA of monkey musk cover between control and full shade reaches from 2014 – 2016					
aov(Monkey musk cover ~ Treatment * Date + Error (Site)					
Error: between					
Treatment	1	2.20	2.20	13.64	<0.01
Residuals	4	0.65	0.02		
Error: within					
Date	15	5.45	0.36	15.86	<0.001
Treatment x date	15	1.73	0.12	5.04	<0.001
Residuals	60	1.38	0.02		

D. Repeated measures ANOVA of watercress cover between control and full shade reaches from 2014 – 2016

aov(Watercress cover ~ Treatment * Date + Error (Site)

Error: between					
Treatment	1	3.87	3.87	2.90	0.16
Residuals	4	5.32	1.33		
Error: within					
Date	15	2.86	0.19	5.23	<0.001
Treatment x date	15	1.55	0.10	2.84	<0.001
Residuals	60	2.19	0.04		

Table S3.4. ANOVA output tables for macrophyte cover and height between control and full shade reaches on the date that shade tunnels were removed.

	d.f.	SS	MS	F	P
A. One-way ANOVA of macrophyte cover on date shade tunnels were removed					
aov(Macrophyte cover ~ Treatment)					
Treatment	1	0.31	0.31	11.5	<0.01
Residuals	4	0.11	0.03		
B. One-way ANOVA of macrophyte height on date shade tunnels were removed					
aov(Macrophyte height ~ Treatment)					
Treatment	1	6.39	6.39	5.22	0.08
Residuals	4	4.90	1.22		
C. One-way ANOVA of monkey musk cover on date shade tunnels were removed					
aov(Monkey musk cover ~ Treatment)					
Treatment	1	0.17	0.17	4.10	0.11
Residuals	4	0.17	0.04		
D. One-way ANOVA of watercress cover on date shade tunnels were removed					
aov(Watercress cover ~ Treatment)					
Treatment	1	0.27	0.27	2.29	0.21
Residuals	4	0.47	0.12		

Plate 4. **Top,** installation of large-scale weed mat and polythene shading. **Centre,** recording a video about the role of science in primary industries for the Ministry for Primary Industries. **Bottom,** removal of large-scale weed mat and polythene shading.

Chapter 4:
Controlling macrophyte growth in small agricultural streams: rethinking the role of shade by riparian buffers

Introduction

Agricultural intensification and other human actions have resulted in extreme channelisation of streams and rivers worldwide, which has generated widespread losses of freshwater biodiversity (Bączyk et al. 2018). The practices of stream channelisation and wetland drainage have been widely used in New Zealand, resulting in profound impacts on water quality, aquatic habitats and invertebrate and fish communities (Collier et al. 1995; Quinn 2000). The clearing of stream bank vegetation reduces organic matter entering the stream, and loss of shading results in an increase in water temperature, nuisance plant growth and altered dissolved oxygen regimes (Barling and Moore 1994; Davies-Colley et al. 2009; Jowett et al. 2009). Furthermore, the introduction of livestock combined with intensification of agricultural land use has resulted in increased runoff, stock trampling and erosion, causing higher levels of suspended sediment and turbidity (Osborne and Kovacic 1993). Higher dissolved nitrogen, phosphorus and faecal indicator bacteria concentrations are typically found in pasture streams compared to those flowing through native forest (Parkyn and Wilcock 2004). Many waterways in lowland regions are now modified drains, which are often considered to have poor biological diversity and little ecological value due to the land-use changes that have occurred within their catchments (Collier et al. 1995; Greenwood et al. 2012; Burdon et al. 2013; Graham et al. 2015). For example, one study reported that in catchments where at least 30% of a catchment has been converted to agriculture, freshwater invertebrate communities can shift from sensitive clean water taxa to pollution tolerant species (Storey and Cowley 1997).

Aquatic macrophytes are present in many rivers and streams, and can provide important services including cycling nutrients, providing habitat for fish and invertebrates, and re-oxygenation of water (Dawson and Haslam 1983; Sand-Jensen and Mebus 1996; Collier et al. 1999; Fleming and Dibble 2014). However, during summer months, excessive macrophyte growth, especially of introduced species, can cause significant issues in agricultural waterways. These excessive macrophytes can reduce water flow, impede drainage functions causing flooding of adjacent land, increase sediment deposition, and alter invertebrate and fish communities (Fox 1992; Collier et al. 1999; Wilcock et al. 1999; Champion and Tanner 2000; Duggan et al. 2002; Bączyk et al. 2018). Due to these negative impacts, management of aquatic macrophytes is commonly undertaken (Fox 1992). The management approach of removing aquatic macrophytes completely from a system alters the ecological balance and thus negatively impacts the ecosystem health of agricultural streams. As a result, there is growing recognition of the need for alternative management regimes. One opportunity may be to re-think the way in which riparian buffers and stream restoration tools are being implemented and improve existing best practice.

The three main management strategies conventionally used to reduce macrophyte biomass in small agricultural streams are mechanical clearance, chemical spray and hand weeding (James 2011; Bączyk et al. 2018). Mechanical clearance usually involves the use of a bankside digger with scoop bucket to physically remove macrophytes and sediment (James 2011). The disturbance caused by mechanical clearance resets macrophyte succession, which can also trigger plant regrowth (Zehnsdorf et al. 2015). Mechanical clearance can be destructive, damaging the stream bed and banks, sediment can be resuspended and released downstream, invertebrates and fish can become caught up and removed in weed biomass and macrophyte fragments can be spread downstream (Hudson and Harding 2004; James 2011; Greer et al. 2012; Zehnsdorf et al. 2015). On occasion chemical sprays are used by applying herbicides (often glyphosate) to kill emergent macrophytes without removing them from the stream (James 2011). Chemical spraying is less physically damaging to the stream and plant dieback can be significant but usually takes time (weeks or months) to occur (James 2011; de Winton et al. 2013). However, there are concerns over the toxicity of chemicals in the environment and oxygen depletion caused by decomposing plants (Jewell 1971; Brooker and Edwards 1975; Young et al. 2004; James 2011). In contrast, hand weeding involves physically cutting macrophyte stems using a sickle or scythe or full removal of all plant and root material. Hand weeding is less ecologically damaging than mechanical clearance and chemical spraying, however it is hugely labour intensive and consequently a much more expensive macrophyte control technique. These three macrophyte management strategies are all short-term solutions, and multiple management actions are often required within a macrophyte growing season (Hudson and Harding 2004; James 2011).

Surprisingly, there is a lack of research on the effectiveness and impacts of these techniques as macrophyte control tools. In several small-scale nuisance macrophyte control trials, Collins et al. (2018a) found that intensive hand weeding, weed mat and full shading (with ca. 80 % light reduction across the channel) were effective control treatments. The intensive hand weeding treatment used was different to hand weeding used by management authorities (typically involving cutting macrophytes back with a sickle or scythe), and involved removing all visible plant and root materials by hand. Macrophyte cover was reduced to < 5 % (compared to 37 % in the untreated control) using this technique (Collins et al. 2018a). Intensive hand weeding is a very labour-intensive control method, and requires ongoing maintenance; however, it has been used to successfully control localised nuisance macrophytes or in conjunction with other management techniques (de Winton et al. 2013; Bellaud 2014).

Another potential method of controlling macrophytes is to use weed mat, or benthic barriers. Plastic sheeting or biodegradable matting have been successful in controlling submerged macrophytes in lakes (Ussery et al. 1997; Caffrey et al. 2010). Benthic barriers have been used in multiple studies, successfully controlling unwanted species including *Lagarosiphon major* (Caffrey et al. 2010), *Najas marina* spp. *intermedia* and *Elodea nuttallii* (Hoffmann et al. 2013) and *Myriophyllum spicatum* (Laitala et al. 2012). However, we were unable to find any published information on the use of weed mat on stream banks to control aquatic macrophytes. We hypothesised that weed mat would be effective at controlling macrophytes which have their roots growing from stream banks by smothering established seedlings and creating a barrier to prevent further plant establishment. We believed that this bank-based control would be effective, given dominant species were the introduced sprawling emergent species *Erythranthe guttata* (monkey musk) and *Nasturtium microphyllum* (watercress), which establish roots in the stream bank and then growth extends out across the waterway. At a small-scale, we demonstrated that weed mat is effective at reducing emergent bank-based macrophyte cover to < 5 %, compared to 37 % in the untreated control (Collins et al. 2018a).

The biomass of most aquatic plants can be reduced by reinstating sufficient shading. Collins et al. (2018a) showed in small scale trials that shading (providing ca. 80 % light reduction) reduced macrophyte cover to 17 % and macrophyte height to < 3 cm. Light strongly affects macrophyte growth (Dawson and Haslam 1983), and is arguably the factor that controls their distribution and abundance at a small scale (Bunn et al. 1998). Due to large-scale land clearance in New Zealand, there are very few lowland streams that are substantially shaded by plants (Champion and Tanner 2000). Where streams are highly shaded, there are few or no macrophytes present, because only a few species are adapted to grow in forested streams (Champion and Tanner 2000; Bowden et al. 2007).

Riparian planting is a commonly used management strategy to restore riparian function and buffer aquatic systems from their surrounding land use (McKergow et al. 2016). Riparian planting can counter some of the negative impacts by providing shade to regulate stream temperature, filtering surface runoff, stabilising stream banks, providing organic matter inputs and habitat for fish and invertebrates, and reducing peak flows (Collier et al. 1995; Fennessy and Cronk 1997; Parkyn et al. 2000; Parkyn et al. 2003). While restoration of riparian planting has been promoted as a means of shading to regulate stream temperatures, it is not often thought of as a means of nuisance macrophyte control.

In this paper, we investigate and demonstrate how traditional riparian buffer design could be improved upon, to result in additional benefits to aquatic systems by controlling nuisance aquatic macrophytes. In Canterbury, riparian buffers generally include a fencing setback of ~ 5 m to exclude stock, combined with planting of 1 – 2 rows of native *Carex* (sedges) which grow to overhang the waterway. A survey of 88 small Canterbury agricultural streams undertaken by Renouf and Harding (2015) found that 65% of riparian buffers were less than 5 m in width, and less than 20% of buffers were > 10 m. They also found that riparian buffers were dominated by exotic pasture grass species, which are unlikely to be providing any shade across the stream (Quinn 2003; Renouf and Harding 2015). If space permits, further rows of flax, native trees and shrubs, are sometimes planted. Several planting guides for riparian restoration in Canterbury have been developed, offering guidance and recommending suitable species to plant (Christchurch City Council 2005; Environment Canterbury Regional Council 2011; DairyNZ 2014). However, with the introduction of centre-pivot irrigation systems, often plant selection is limited to species which grow below rotating sprinklers at 2–3 m.

Widespread clearance of native vegetation and land-use conversion to agriculture have been undertaken since human settlement in New Zealand, such that indigenous forest now covers only 24 % of the total land area (Ewers et al. 2006). In comparison, 40 % of the land area has been converted to exotic pasture grass grazed by ruminant animals including cattle, sheep and deer (Scarsbrook et al. 2016). Farming is now the most common land use in the middle to lower catchments of many New Zealand rivers (Storey and Cowley 1997; Quinn 2000). A major threat to freshwater systems is the expansion and intensification of pastoral dairy farming. Sheep numbers peaked at 70 million in the early 1980s, followed by significant intensification of farming which resulted in declining numbers of sheep (50 million in 1994 to 30.8 million in 2013) and beef cattle (5 million in 1994 to 3.7 million in 2012) but increasing numbers of dairy cattle (3.8 million in 1994 to 6.5 million in 2012) (Scarsbrook et al. 2016). The increase in dairy farming is especially pronounced in the South Island of New Zealand, with stock numbers increasing from 0.5 million in 1994 to 2.5 million in 2012 (Scarsbrook et al. 2016). This large-scale dairy conversion has been accompanied by increased irrigation, fertiliser application, effluent disposal and the introduction of nitrogen-fixing plants (Willis 2001; Clark et al. 2007; Baskaran et al. 2009) and has resulted in marked changes to freshwater ecosystems. Many waterways in lowland regions are now modified drains, which are often considered to have poor biological diversity and little ecological value. In contrast with this perception, drains provide habitat, support invertebrate and fish species and are often the last remnants of substantial wetlands that historically covered New Zealand’s lowland areas (Young et al. 2004; James 2011).

The aim of this study was to evaluate the effectiveness and suitability of three macrophyte control techniques in lowland agricultural drains; intensive hand weeding, weed mat and full shading (100%) to control aquatic macrophytes at a large scale (up to 400 m reaches). We then use these findings to show how rethinking traditional riparian buffer design could result in additional benefits to aquatic systems by controlling aquatic macrophytes.

Materials and methods

Study sites

This study was carried out on three small streams – Harris Drain near Ashburton and Todds Drain and South Brook, both in Rangiora in the Canterbury Region, South Island, New Zealand. Harris Drain is a spring- and surface-fed stream that flows through cropping land then discharges directly to the coast south of Ashburton. Todds Drain is a roadside drain that flows through rural and industrial land use, whereas South Brook flows alongside a reserve. Dominant macrophytes in all streams were the introduced emergent species *Erythranthe* *guttata* and *Nasturtium microphyllum*, with other species present at lower abundance (Table 4.1). The streams all had a wetted width of 2–2.5 m, depth of 20–40 cm, discharge of 0.04–0.08 m^3/s and cobble substrate covered with a layer of fine sediment. The experimental reach of Todds Drain flowed NNE to SSW, and Harris Drain and South Brook flowed NNW to SSE.

Table 4.1: Macrophyte species observed in each stream.

Species name	Common name	Harris Drain	Todds Drain	South Brook
Erythranthe guttata	Monkey musk	✓	✓	✓
Nasturtium microphyllum	Watercress	✓	✓	✓
Glyceria fluitans	Floating sweetgrass	✓	✓	✓
Callitriche stagnalis	Water starwort	✓	✓	✓
Myosotis sp.	Water forget-me-not	✓	✓	
Veronica anagallis-aquatica	Water speedwell	✓	✓	
Potamogeton crispus	Curly pondweed	✓	✓	
Juncus articulatus	Jointed rush	✓	✓	
Myriophyllum aquaticum	Parrot's feather	✓		
Ranunculus trichophyllus	Water buttercup	✓		
Elodea canadensis	Canadian pondweed		✓	

Harris Drain, Todds Drain and South Brook are highly modified waterways and have been actively managed by local water management authorities over the last several decades. The removal of excessive nuisance macrophytes in these streams has been undertaken using a combination of mechanical excavation and chemical spray using glyphosate. Prior to our macrophyte trial, the stream banks at Harris Drain were rebattered to create gently sloping banks and stop bank collapse, and the riparian zone was fenced to exclude livestock and planted with two rows of native sedges including *Carex secta*, *Carex virgata* and *Cyperus ustulatus* and a further row of low growing native shrubs.

We conducted two trials: a reach-scale trial of intensive hand weeding, polythene shading and weed mat in Harris Drain, and a large-scale trial of weed mat and intensive hand weeding in Todds Drain and South Brook.

Reach-scale trials

Our trial investigated the effectiveness of intensive hand weeding, weed mat and shading to reduce aquatic macrophyte biomass over 50-m reaches at Harris Drain. Here, each treatment (Table 4.2) was replicated three times down a 600-m stretch. Previous research in this waterway showed that upstream macrophyte manipulations had no effect on downstream macrophyte growth and biomass (CAREX unpublished data). Therefore, we were not concerned that treatments along a length would affect each other. Treatments were randomly assigned to each of the reaches. Different treatments targeted different types of macrophytes: intensive hand weeding and polythene shading targeted submerged bed macrophytes, whereas weed mat targeted sprawling emergent bank macrophytes. Consequently, we ran two separate analyses: a sprawling emergent bank trial which compared weed mat treatment to untreated control reaches; and a submerged bed trial, comparing intensive hand weeding and polythene shading to untreated control reaches.

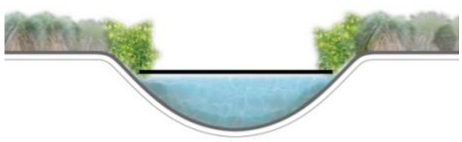
Reach-scale sprawling emergent bank trial

Prior to treatment in December 2015, macrophyte cover was measured in the control and weed mat treatments by randomly placing 10 quadrats (30 cm x 30 cm) on the stream bank within each reach. In each quadrat the percentage cover of each macrophyte species was estimated visually. The weed mat was installed in December 2015 (Table 4.2). Macrophyte percentage cover was then re-measured after one growing season (April 2016), and two growing seasons (April 2017). Macrophyte percentage cover from the 10 quadrats within each reach were combined for analysis.

Reach-scale submerged bed trial

Prior to treatment in December 2015, macrophyte cover was measured in the control, intensive hand weeding and polythene shading treatments by randomly placing 10 quadrats (30 cm x 30 cm) in the wetted cross section within each reach. In each quadrat the percentage cover of each macrophyte species was estimated visually. The hand weeding treatment was undertaken and polythene shading installed in December 2015 (Table 4.2). Macrophyte percentage cover was then re-measured after one growing season (April 2016), and two growing seasons (April 2017). Macrophyte percentage cover from the ten quadrats within each reach were combined for analysis.

Table 4.2. Experimental treatments investigated (each with three replicates) in the reach-scale sprawling emergent bank submerged bed trials.

Treatment	Description	Target macrophytes
Control	No treatment within 50 m reach	
Weed mat	50 m lengths of 0.91 m wide plastic woven 100 gsm weed mat (Egmont Commercial) were installed on both stream banks and pegged down with plastic pegs	Sprawling emergent bank 
Intensive hand weeding	All visible submerged plant and root materials were removed within 50 m reach	Submerged bed 
Polythene shading	50 m lengths of 250 μm black polythene (Egmont Commercial) suspended over water surface	Submerged bed 

Large-scale trials

Large-scale weed mat trial

At Todds Road, 400 m of both stream banks were covered with 1.83 m plastic woven 100 gsm weed mat (Egmont Commercial) and pegged down with plastic pegs. Weed mat was installed in December 2015. A control reach was left untreated upstream of the treatment reach. Within the weed mat and control treatments, ten permanent macrophyte assessment transects were established across the reach. In each assessment transect, macrophyte species were identified and their height above the water surface was measured with a ruler at every 10 cm across the transect, thus each transect had about 15 – 20 measurements. Macrophyte assessment transects were measured after one growing season (April 2016) and two growing seasons (April 2017). Each transect was used as a replicate in analysis.

Large-scale hand weeding trial

At South Brook, a 200-m length of stream was hand weeded, and a 200-m reach was left untreated downstream of the hand weeded reach as a control reach. Hand weeding was undertaken in February 2016. Within the hand weeding and control treatments, five permanent macrophyte assessment transects were established. In each assessment transect, macrophyte species were identified and their height above the water surface was measured with a ruler at every 10 cm across the transect, thus each transect had about 15 – 20 measurements. Macrophyte assessment transects were measured immediately post-treatment in February 2016, and three months post treatment (May 2016). Each transect was used as a replicate in analysis.

Statistical analyses

For each trial, changes in macrophyte cover between treatments over time were assessed using repeated measures ANOVA with date and treatment as factors, a treatment x date interaction and treatment reach as an error term in base R (R Core Team 2014). Post-hoc comparisons of means were made using Tukey HSD tests.

All statistical analyses were undertaken using R statistical software version 3.1.2 (R Core Team 2014) and macrophyte percent cover values were normalised by arcsine square-root transformation prior to analyses (Zar 2009).

Results

Reach-scale trials

Reach-scale sprawling emergent bank trial

In the reach-scale sprawling emergent bank macrophyte trial, *post hoc* Tukey tests showed that there was no difference in macrophyte cover between reaches prior to treatment in December 2015. There was a statistically significant difference between the weed mat and control treatments over time (treatment x time interaction, $F_{2,8} = 9.67$, $P < 0.01$; Figure 4.1, Table S4.1A). After one growing season in April 2016, macrophyte cover in the weed mat reaches was 3 %, compared to cover in control reaches at 58 % and pre-treatment weed mat reaches at 62 % (Figures 4.1 & 4.2). After two growing seasons in April 2017, macrophyte cover in the weed mat reaches remained suppressed at 6 %. Macrophyte cover in the control reaches was also reduced to 12 % after two growing seasons, however it was not statistically different from either the weed mat reaches, or the control after one growing season (Figure 4.1).

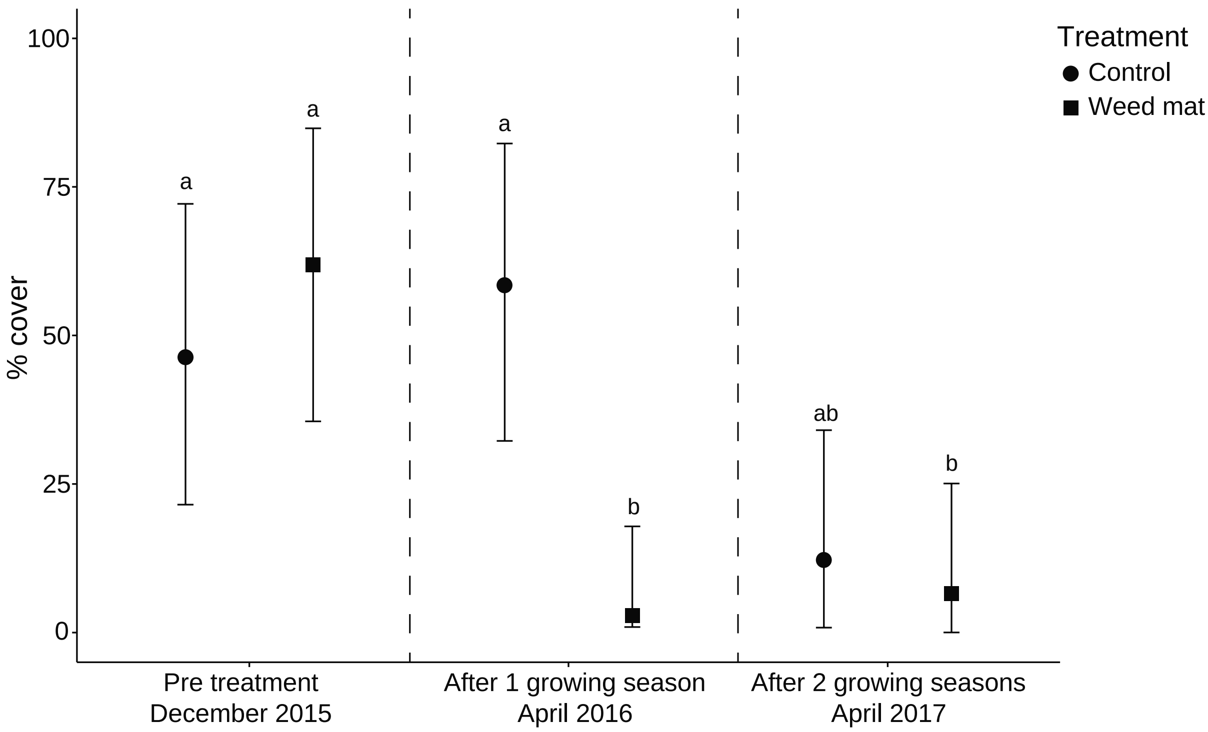


Figure 4.1. Mean (± 95 % CI) macrophyte cover illustrating the effectiveness of weed mat relative to control reaches for sprawling emergent bank macrophytes over two macrophyte growing seasons (n = 3). Letters above error bars indicate statistically significant differences between individual treatments at peak macrophyte cover.

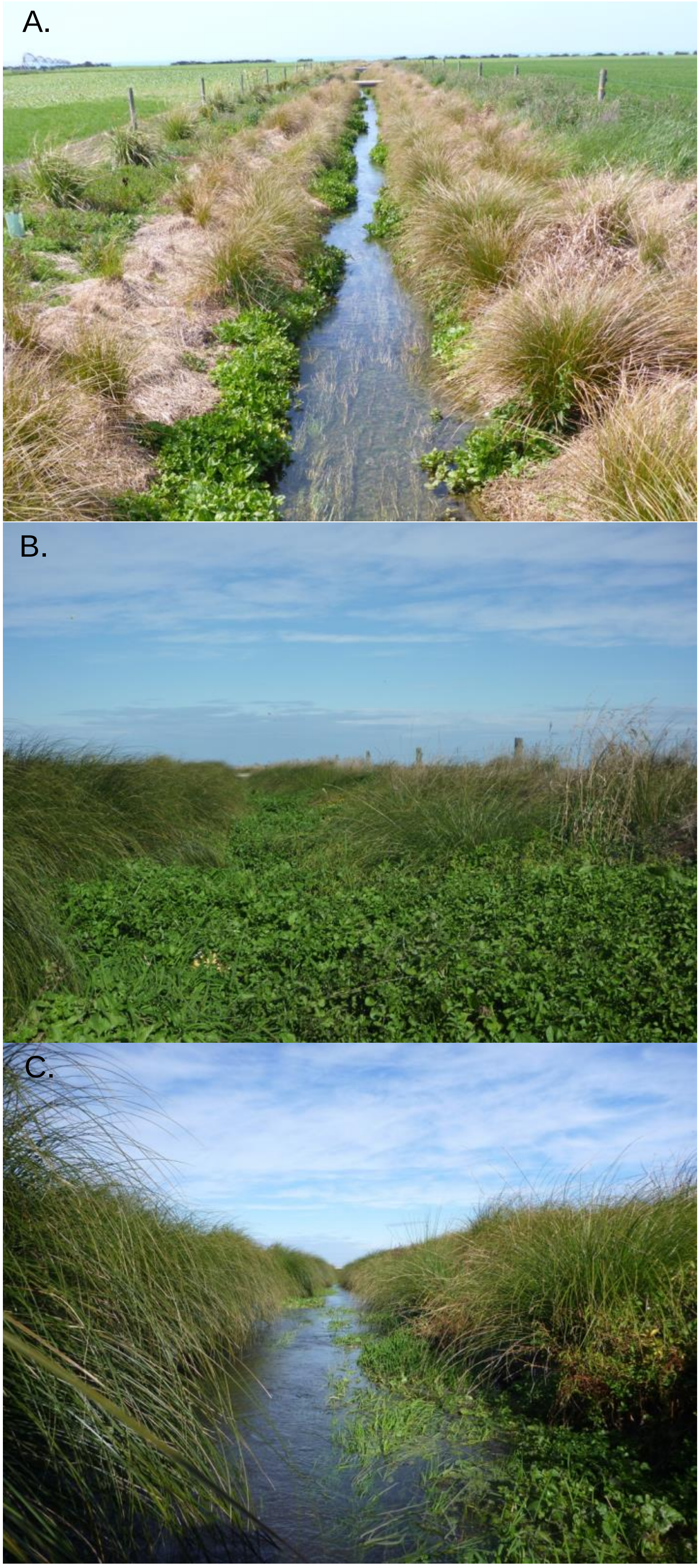


Figure 4.2. Photographs of treatment reaches in reach-scale trials for management of sprawling emergent bank macrophytes including: **A**, Control reach pre-treatment in December 2015. **B**, Control reach after one growing season in April 2016. **C**, Weed mat treatment reach after one growing season in April 2016.

Reach-scale submerged bed trial

In the reach-scale submerged bed macrophyte trial, *post hoc* Tukey tests showed that there was no difference in macrophyte cover between reaches prior to treatment in December 2015. There was a statistically significant difference between the hand weeding, polythene shading and control treatments over time (treatment x time interaction, $F_{4, 12} = 23.54$, $P < 0.001$; Figure 4.3, Table S4.1B). After one growing season in April 2016, macrophyte cover in the polythene shading reaches was absent (0 %) compared to control (67 %) and hand weeding (32 %) reaches (Figures 4.3 & 4.4). However, large scale hand weeding of submerged macrophytes was not an effective macrophyte reduction technique and cover had returned to 63 % after two growing seasons. After two growing seasons in April 2017, macrophyte cover in the polythene shading reaches remained suppressed. Macrophyte cover in the control reaches increased from pre-treatment over two growing seasons to 81 % in April 2017 (Figure 4.3).

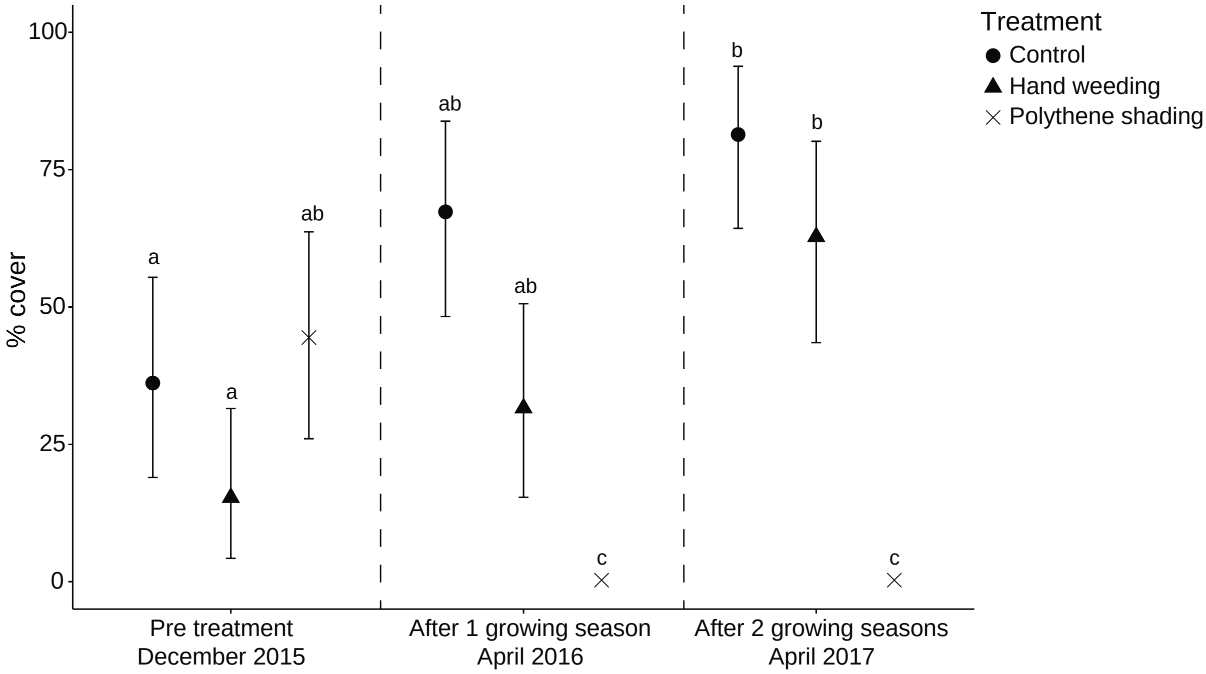


Figure 4.3. Mean (± 95 % CI) macrophyte cover illustrating the effectiveness of hand weeding and polythene shading relative to control reaches for bed macrophytes over two macrophyte growing seasons ($n = 3$). Letters above error bars indicate statistically significant differences between individual treatments at peak macrophyte cover.

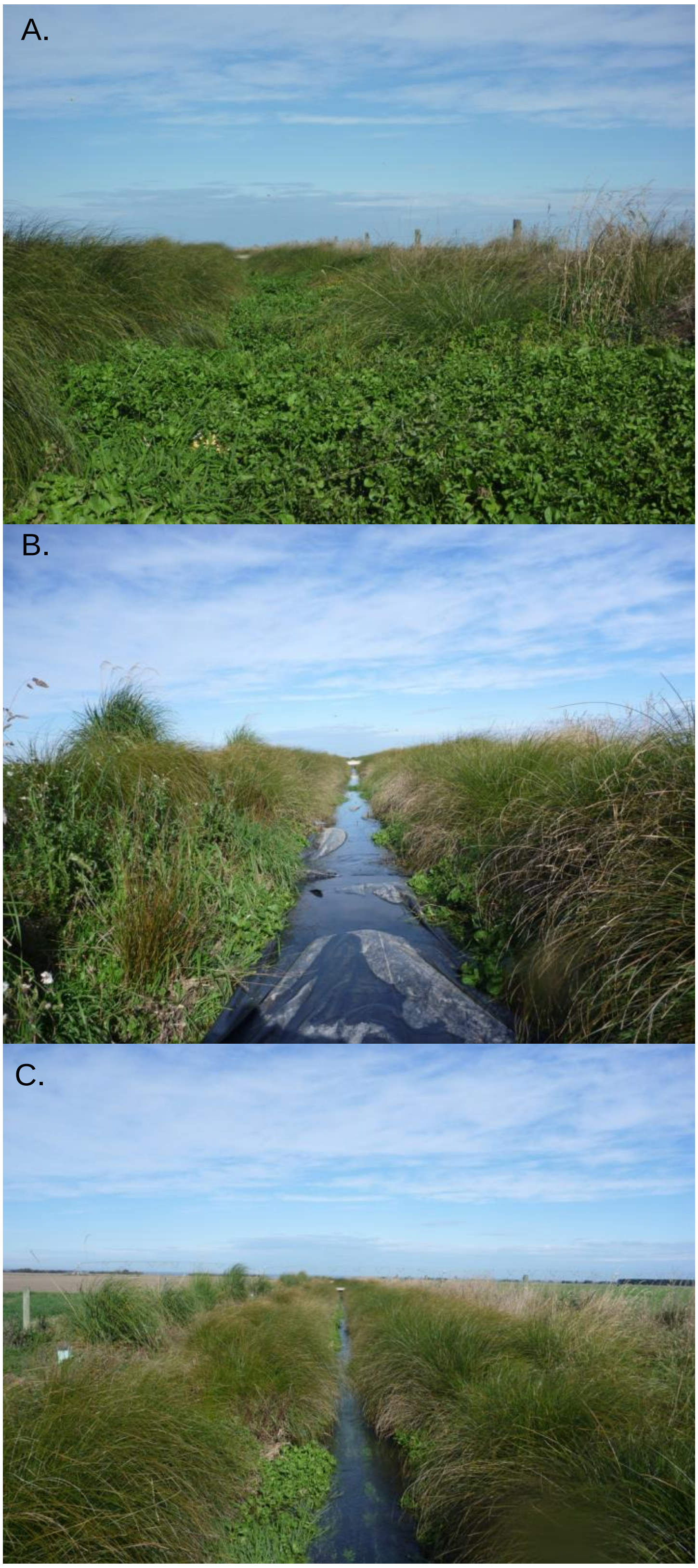


Figure 4.4. Photographs of treatment reaches in reach-scale trials for management of submerged bed macrophytes including: **A**, Control reach after one growing season in April 2016. **B**, Polythene shading treatment reach after one growing season in April 2016. **C**, Hand weeding treatment reach after one growing season in April 2016.

Large-scale trials

Large-scale weed mat trial

In the large-scale weed mat trial, average macrophyte cover was 80 % during the pre-treatment sampling in December 2015, however after treatment there was a statistically significant difference between the control and weed mat treatments (treatment effect, $F_{1,36} = 98.29$, $P < 0.001$; Figure 4.5, Table S4.1C). *Post hoc* Tukey tests showed after one growing season in April 2016, macrophyte growth was greatly reduced in the weed mat reach (45 %) compared to the control (98 %, Figures 4.5 & 4.6). After two growing seasons in April 2017, macrophyte cover in the weed mat reach was further reduced to 12 %. There was no difference in macrophyte cover in the control reach between the two growing seasons (Figure 4.5).

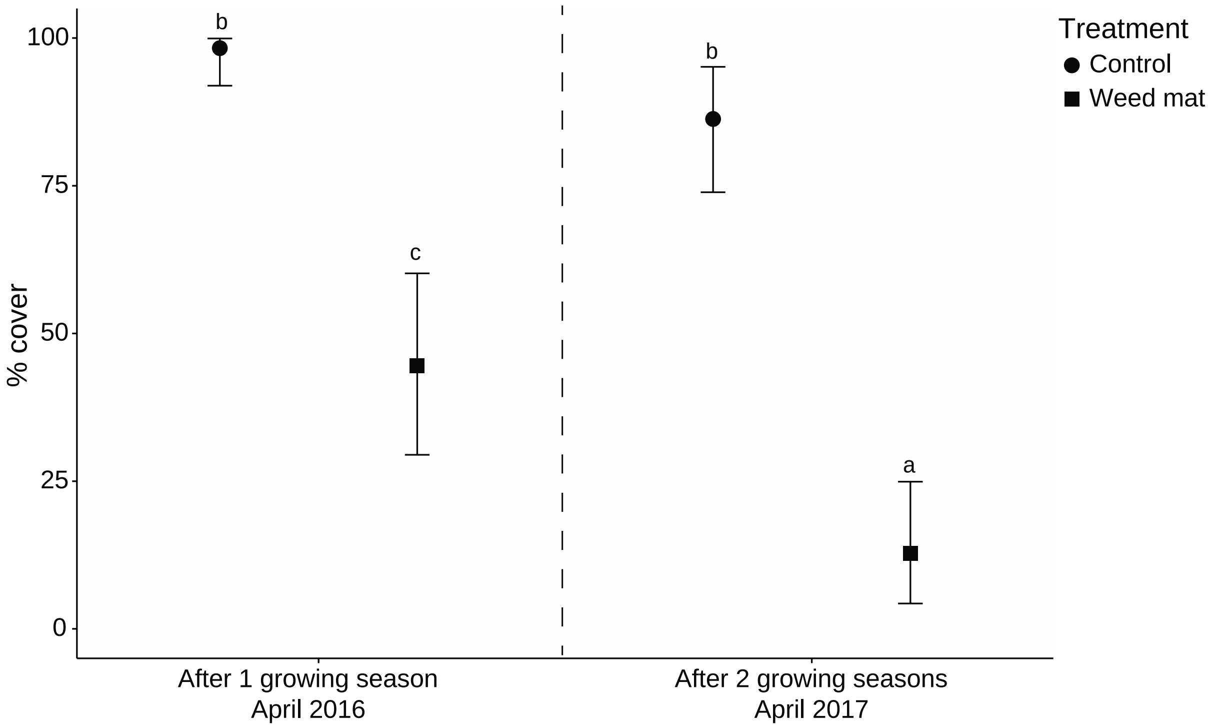


Figure 4.5. Mean (± 95 % CI) macrophyte cover illustrating the effectiveness of weed mat in treated reaches relative to control reaches over two macrophyte growing seasons (n = 10). Letters above error bars indicate statistically significant differences between individual treatments at peak macrophyte cover.

Figure 4.6. Photographs of treatment reaches in large-scale trials of weed mat for management of sprawling emergent bank macrophytes after one growing season in April 2016: **A**, Control reach. **B**, Weed mat treatment reach.

Large-scale hand weeding trial

In the large-scale hand weeding trial, average macrophyte cover was 85 – 95 % in South Brook during the pre-treatment sampling in February 2016, however after hand weeding there was a significant difference between the control and hand weeding treatments over time (treatment x time interaction, $F_{1,1} = 569.7$, $P < 0.001$; Figure 4.7, Table S4.1D). *Post hoc* Tukey tests showed on the day of hand weeding, macrophyte cover in the hand weeding reach was completely removed compared to the control at 91 % cover (Figures 4.7 & 4.8). Nevertheless, after one growing season in April 2016, macrophytes had rapidly recovered in the hand weeded reach to 92 % cover, and macrophyte cover was slightly higher than in the control reach at 82 % (Figure 4.7).

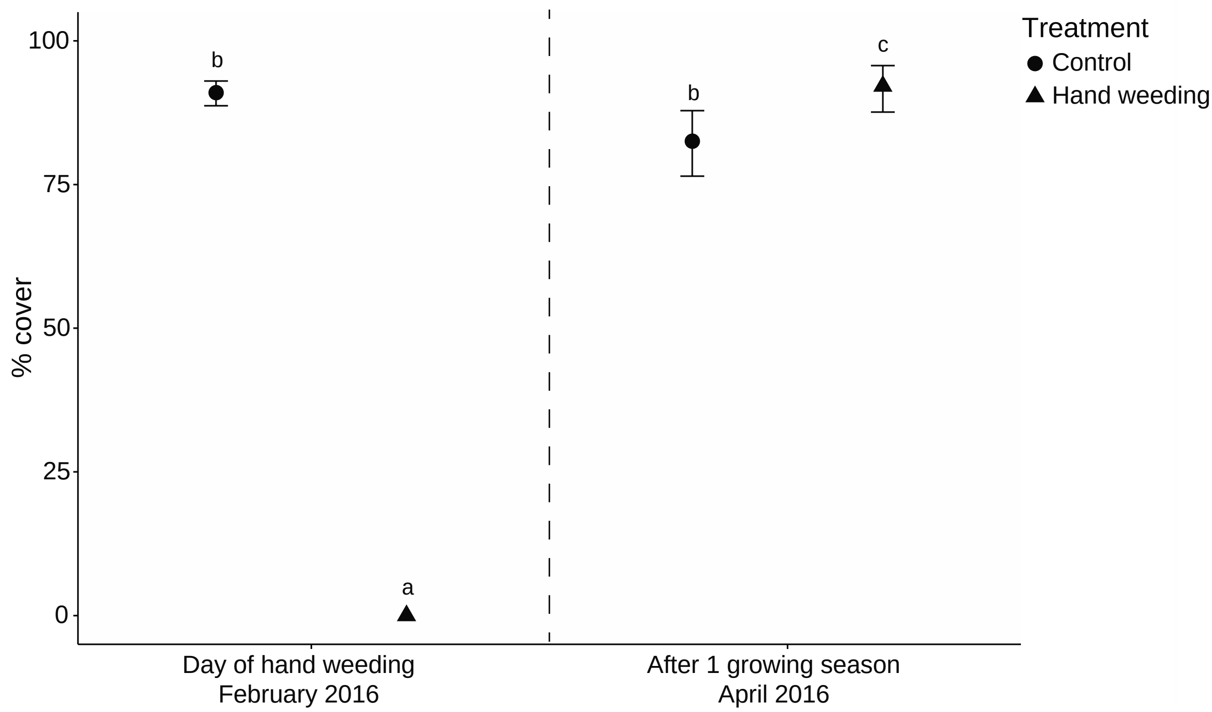


Figure 4.7. Mean (± 95 % CI) macrophyte cover illustrating the effectiveness of hand weeding relative to control reaches over one macrophyte growing season (n = 5). Letters above error bars indicate statistically significant differences between individual treatments at peak macrophyte cover.

Figure 4.8. Photographs of treatment reaches in large-scale trial of hand weeding for management of sprawling emergent bank macrophytes including: **A**, Pre-treatment in February 2016. **B**, Hand weeding treatment reach immediately after the weeding treatment in February 2016. **C**, Hand weeding treatment reach after one growing season in May 2016.

Discussion

The current techniques employed to manage nuisance aquatic macrophytes can be ecologically damaging, costly or only provide short-term control. The aim of this study was to evaluate the effectiveness and suitability of three macrophyte alternative control techniques: intensive hand weeding, weed mat and polythene shading to control aquatic macrophytes at reach (50 m hand weeding, weed mat and polythene) and larger (up to 400 m hand weeding and weed mat) scales. These three techniques were shown to be effective in small-scale trials (Collins et al. 2018a).

Hand weeding

Hand weeding is used on occasions as a macrophyte management technique in waterways globally (Hinds Drains Working Party 2016; Bączyk et al. 2018). In Canterbury, the rural Ashburton Hinds Drainage District estimate that less than one percent of drains within the rating district are cleared using hand weeding (Hinds Drains Working Party 2016); however, in central Christchurch City this percentage is much higher. Each summer-autumn, the Christchurch City Council hires teams of labourers who physically cut and remove macrophytes from waterways. This makes hand weeding a very labour-intensive and costly control technique. Therefore, hand weeding is typically used where mechanical clearance or chemical application are unsuitable. These include areas where bank-side access for diggers is limited, there is opposition to the use of herbicides, or herbicide use is inappropriate due to overhanging riparian planting. Macrophyte recovery following hand weeding can occur within weeks to months, with plants re-establishing from stems that were left intact, cut fragments that were not removed from the stream, and the seed bank. However, with regular maintenance, small-scale hand weeding can be an effective macrophyte control technique (Collins et al. 2018a).

Our large-scale intensive hand weeding treatment at South Brook created a physical disturbance of the stream bed and banks, and exposed the bed and banks to more light. This increased light, combined with the bare substrate provided an ideal habitat for macrophyte regeneration from the residual seed bank. Additionally, macrophyte fragments could have entered the hand weeded reach from upstream sources. In contrast, a slower rate of macrophyte recovery was observed at Harris Drain, where efforts were focussed on removing submerged macrophytes and some shading was provided by the *Carex* plantings in the riparian buffer which grew during the course of the experiment.

While we did not find intensive hand weeding to be an effective large-scale long-term macrophyte control option, hand weeding does have a place in macrophyte control. Intensive hand weeding can be an effective technique for targeted removal of undesirable species while allowing the rest of the community to be left intact (Chisholm 2006; Bellaud 2014). An example of this is the removal of *Trapa natans* (water chestnut) in several New York lakes, where hand weeding efforts have been co-ordinated between management agencies and volunteer efforts (Hummel and Kiviat 2004). Hand weeding has also been used as an incursion response for small infestations of high risk species in New Zealand (P. Champion, personal communication). However, hand weeding is incredibly labour intensive and costly, and to be effective, ongoing surveillance and maintenance is necessary (Hussner et al. 2017).

Weed mat

Significant research effort has gone into the use of weed mat to control submerged macrophytes in lakes (Ussery et al. 1997; Caffrey et al. 2010), including the best way to install weed mat and determining the most suitable material to use. Plastic matting was found to be buoyant, restricting macroinvertebrate and nutrient movement, and reducing oxygen levels (Ussery et al. 1997; Caffrey et al. 2010; Hofstra and Clayton 2012; Hoffmann et al. 2013). Thus, a jute material has been suggested as a preferable alternative as it sinks easily, allows gas and nutrient exchange and allows native macrophyte communities to re-establish through the loose-weave of the matting. The jute matting has an additional benefit of biodegrading in the environment and not requiring removal.

In contrast to this work in lakes, we have been unable to find any published information on the use of weed mat in flowing waters to control aquatic macrophytes. Although several years ago Christchurch City Council did trial unsuccessfully to smother submerged macrophytes in the Avon River (J. Harding, personal communication). In a previous study, we trialled the use of weed mat at a small-scale on stream banks and found it to be an effective macrophyte control of monkey musk and watercress which have roots in the banks and grow out into the waterway (Collins et al. 2018a). In that work we found that some types of weed mat were better than others (Collins et al. 2018a). The biodegradable wool weed mat had a very limited lifespan on stream banks (< 3 months). In contrast, plastic woven weed mat was a more hard-wearing and long-lived product. Additionally, in small plots (2 m x 2 m) there was a large influence of edge effects with plant recolonisation occurring from the edges (Collins et al. 2018a).

At Harris and Todds Drains, reach- to large-scale bank weed mat was an effective control technique over two growing seasons. By using long lengths of weed mat we did not observe any impact of edge effects. At Harris Drain, the stream banks were re-battered and *Carex* were planted prior to the trial commencing in October 2014. Small holes were cut to allow the *Carex* plants to grow through the weed mat. Over two years, the *Carex* seedlings had grown through the weed mat into large plants (1.5 – 2 m) in diameter, providing shading to both the bank and water surface. The use of weed mat also had the added benefit of retaining soil moisture and preventing nuisance terrestrial weeds from establishing in newly planted riparian buffers.

Ideally, a biodegradable material would be an ideal alternative in this situation as it would not require removal once plantings have established, however, we have found that currently available biodegradable materials either break down fast (EcoWool mulch mat, < 3 months) or allow establishment of monkey musk and watercress through coarse weave (coir coconut fibre matting). This leaves the potential for the development of a novel material that has a tight weave and slow breakdown rate.

Polythene shading

Light availability impacts the ability for plants to photosynthesise, thus affecting macrophyte presence and abundance (Dawson and Haslam 1983). In streams flowing through closed forest canopies, there are few macrophytes present (Champion and Tanner 2000; Bowden et al. 2007). Reinstating riparian cover can greatly decrease macrophyte growth in streams (Dawson and Haslam 1983), however, planting is more effective at providing shading in smaller channels (O'Briain et al. 2017; Willis et al. 2017). Our small-scale macrophyte control trial showed shading reduced both macrophyte height and cover (Collins et al. 2018a) and polythene shading was effective for controlling submerged bed macrophytes in Harris Drain.

During the pre-treatment phase at Harris Drain, stream banks were re-battered, grass seed scattered and *Carex* planted to create shading. After the first growing season, bank macrophytes were still growing in the control reaches. After two growing seasons, the *Carex* plants were well established and beginning to provide shading. Sprawling emergent bank macrophyte cover was beginning to decline, as seen in the control reaches of our reach scale trial (Figure 4.1). This suggests that riparian planting can provide sufficient shading to control sprawling emergent bank macrophytes within two years.

We conclude that weed mat can be a practical and effective short-term macrophyte control solution while riparian plantings establish. In situations where riparian planting is being undertaken, and nuisance aquatic macrophytes are a consideration, cutting small holes to allow planting into extended lengths of weed mat as a method to effectively control macrophyte growth in the short term. In the longer term, riparian plants will establish and grow up, providing necessary shade to ensure continued macrophyte control. Consideration should be given to placement of riparian plant species that will grow up to provide shading, these should be sited to provide maximum shade across the stream channel. Another often overlooked factor, is the placement of the bottom row of plants. The bottom row of *Carex* plants should be placed as close to the edge of the stream wetted cross section as is practical, to ensure maximum benefit of shade provided by overhanging riparian vegetation is reached in the shortest possible time. Undertaking riparian restoration in this way will aid the recovery of instream habitat, in addition to the riparian buffer, helping to restore the balance in stream systems and reduce the need for costly management intervention.

Supplement to Chapter 4

Table S4.1. ANOVA output tables for macrophyte cover, **A,** Reach-scale emergent macrophyte control trial to accompany Figure 4.1, **B,** Reach-scale submerged bed macrophyte control trial to accompany Figure 4.3, **C,** Large-scale weed mat macrophyte control trial to accompany Figure 4.5, and **D,** Large-scale hand weeding macrophyte control trial to accompany Figure 4.7.

	d.f.	**SS**	**MS**	**F**	**P**
A. Reach-scale sprawling emergent bank macrophyte control trial					
aov(Macrophyte cover ~ Treatment * Date + Error (Bridge))					
Error: between					
Treatment	1	0.35	0.35	18.88	< 0.05
Residuals	4	0.08	0.02		
Error: within					
Date	2	0.82	0.41	13.63	< 0.01
Treatment x date	2	0.58	0.29	9.67	< 0.01
Residuals	8	0.24	0.03		
B. Reach-scale submerged bed macrophyte control trial					
aov(Macrophyte cover ~ Treatment * Date + Error (Bridge))					
Error: between					
Treatment	2	2.01	1.00	24.66	< 0.01
Residuals	6	0.24	0.04		
Error: within					
Date	2	0.12	0.06	3.60	< 0.05
Treatment x date	4	1.56	0.39	23.54	< 0.001
Residuals	12	0.20	0.02		
C. Large-scale weed mat macrophyte control trial					
aov(Macrophyte cover ~ Treatment * Date + Error (Treatment))					
Error: between					
Treatment	1	5.89	5.89	92.57	< 0.001
Residuals	18	1.15	0.07		
Error: within					
Date	1	0.94	0.94	16.75	< 0.001
Treatment x date	1	0.03	0.03	0.61	0.45
Residuals	18	1.01	0.06		

D. Large-scale hand weeding macrophyte control trial					
aov(Macrophyte cover ~ Treatment * Date + Error (Treatment))					
Error: between					
Treatment	1	1.57	1.57	89.37	< 0.001
Residuals	8	0.14	0.0175		
Error: within					
Date	1	1.68	1.68	114.5	< 0.001
Treatment x date	1	2.49	2.49	169.8	< 0.001
Residuals	8	0.12	0.01		

Plate 5: Two examples of shading provided by riparian planting achieving macrophyte control.

Chapter 5:
General discussion

Aquatic macrophytes provide important functions in fresh waters, but excessive growth of introduced macrophytes in small agricultural streams can have significant negative impacts (Dawson and Hallows 1983; Sand-Jensen and Mebus 1996; Collier et al. 1999; Fleming and Dibble 2014). These impacts include reducing water flow, accumulating sediment, impeding drainage and causing flooding of adjacent land (Fox 1992). Many New Zealand farmers see drainage as the primary function of small agricultural waterways (Hudson and Harding 2004; Greer et al. 2012). Farmers and landowners require these waterways to remove water efficiently and quickly from the farm. During spring and summer, agricultural streams globally can become choked with macrophytes, reducing drainage capability and requiring management to control their growth (Fox 1992).

Large-scale pastoral land use conversion has resulted in extensive stream channelisation and wetland drainage, such that many lowland streams are now modified drains. These drains are perceived to be primarily for removing floodwaters and high flows as efficiently as possible and often considered to have little ecological value. Contrary to this, studies in both New Zealand and overseas have shown that drains can contain significant aquatic biodiversity (Armitage et al. 2003; Herzon and Helenius 2008; Sinton 2008; Simon and Travis 2011). Additionally, drainage networks create connectivity through modified landscapes, and often may flow into highly valued streams and receiving environments (Sinton 2008).

The aims of my thesis were to understand the factors that influence macrophyte biodiversity, abundance and biomass in agricultural streams in Canterbury, and to evaluate the effectiveness of alternative practical macrophyte control options.

My thesis was undertaken as part of the Canterbury Waterway Rehabilitation Experiment (CAREX, www.carex.org.nz), a project within the Freshwater Ecology Research Group at the University of Canterbury. This long-term, collaborative project identified that building strong partnerships with landowners, stakeholders and local management agencies was key to achieving freshwater restoration success. To be successful, solutions to the key multiple stressors impacting agricultural streams, including excessive aquatic macrophytes, high nutrients, high deposited fine sediment and low aquatic biodiversity needed to be co-developed with local stakeholders. The project aimed to communicate research findings in an understandable and accessible way, through handouts and quarterly newsletters to landowners, stakeholders, management agencies and other interested parties (Collins et al. 2018b; Harding et al. 2018). The nine streams the project focussed on were flowing through operational farms, and any proposals needed to be practical and ensure farming practices could continue with minimal disruption.

Understanding factors that influence macrophyte diversity and abundance

I undertook a region-wide survey (Chapter 2), where I proposed and tested a conceptual model (Figure 2.1, also reprinted as Figure 5.1) of factors that influence macrophyte diversity and abundance and the spatial scales they operate at. In my survey I was able to test a number of components (but not all) of my conceptual model. The survey suggested that at the reach scale, the natural disturbance regime (e.g. flood events) is likely the key factor limiting macrophyte growth compared to stream shade at the patch scale. Nutrient concentrations did not appear to limit macrophyte growth. Improving our understanding of these factors that influence macrophyte biomass is helpful in terms of informing alternative management regimes to manage excessive growth. The survey findings suggested that physical disturbance, artificial high flow events and shading would provide effective macrophyte management tools. I believe the conceptual model I created and tested is useful and applicable to other geographic regions, and across macrophyte species. However, further research is needed to test the relationships between each factor and the response of macrophytes.

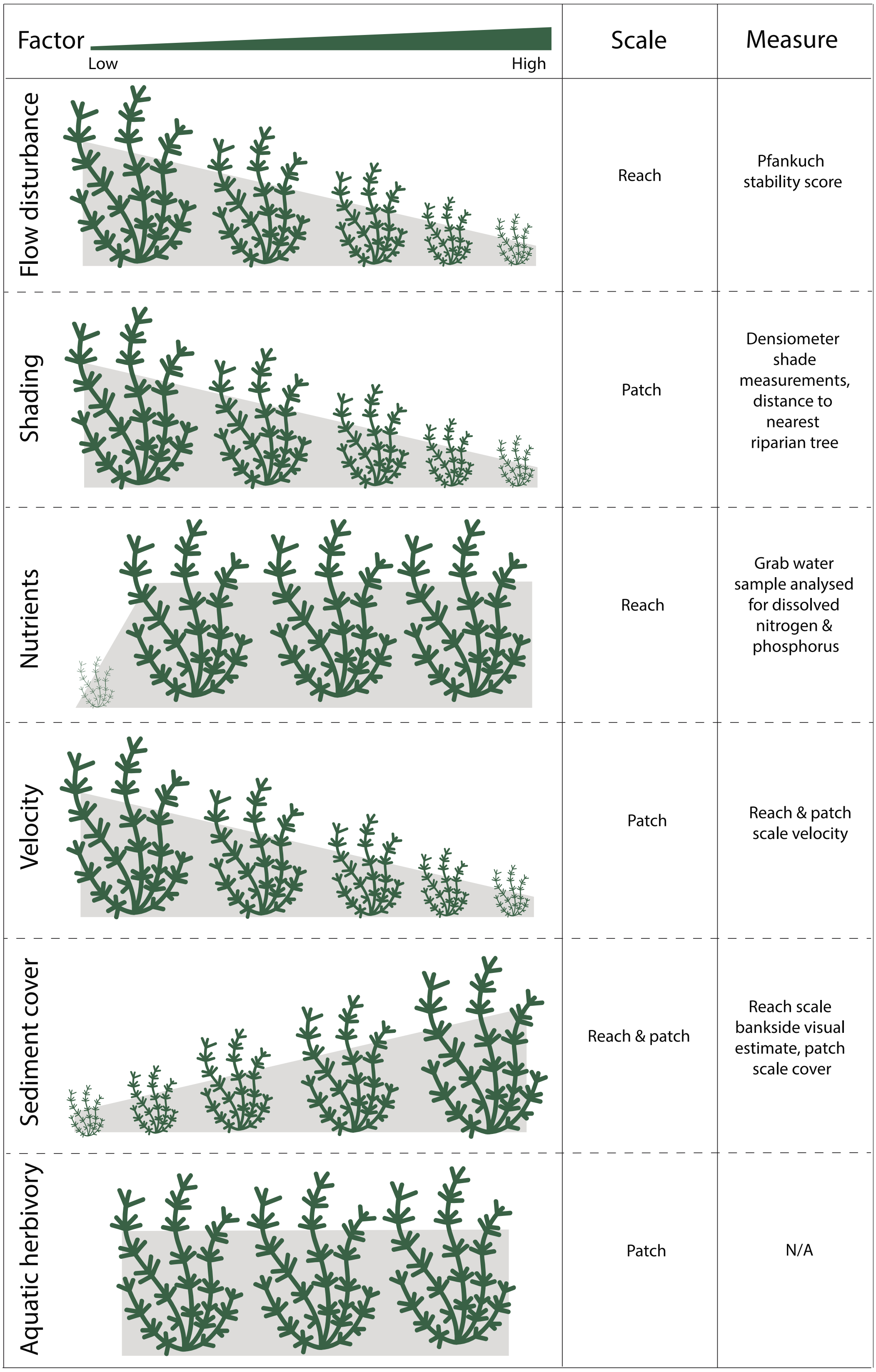


Figure 5.1. Conceptual model of factors likely to affect macrophyte abundance in small agricultural streams in New Zealand and the scale which they operate at. Factors increase from left to right along the x-axis, with larger plant sizes indicative of greater macrophyte abundance (Reprinted from Chapter 2, originally appeared as Figure 2.1).

There were several limitations of the survey design. I surveyed sites that were known to have macrophytes present, to understand the factors that drive macrophyte abundance and diversity. The aim of the survey was not to investigate the factors that drive macrophyte presence or absence. As such, some elements of our conceptual model were not tested, namely highly physically-disturbed (flood-prone) streams, which have no macrophytes present. Furthermore, I did not investigate the impact of aquatic herbivory on macrophyte biomass. Kōura (freshwater crayfish), invertebrate grazers, herbivorous fish and aquatic birds can consume macrophytes (Matheson et al. 2012), however, all are uncommon in Canterbury lowland streams. Thus, I assumed any effect of herbivores on macrophyte biomass in these Canterbury waterways was minimal.

Conventional macrophyte control techniques

Conventional macrophyte control techniques in flowing waters include mechanical clearance, chemical sprays and hand weeding (James 2011). These techniques can be costly and potentially detrimental to the aquatic communities that inhabit them. For example, mechanical clearance can release sediment, over-steepen banks, spread fragments downstream, and fish and invertebrates can be removed from the channel, and chemical control releases herbicide to the environment and can deplete oxygen levels (Hudson and Harding 2004; James 2011; Greer et al. 2012; Zehnsdorf et al. 2015). Additionally, their effectiveness is relatively short-lived, and results are generally not sustained beyond a single growing season.

I believe part of the fundamental issue and acceptance of the impacts of conventional macrophyte control techniques comes from labelling these streams as "drains". Part of the reason for this classification is how different types of waterways are defined in New Zealand legislation and planning documents. The Resource Management Act (RMA, 1991) is the primary piece of legislation surrounding the environment in New Zealand, and it sets the overall framework for resource management. The RMA defines a river as *"a continually or intermittently flowing body of fresh water; and includes a stream and modified watercourse; but does not include any artificial watercourse (including an irrigation canal, water supply race, canal for the supply of water for electricity power generation, and farm drainage canal)"*. In the Canterbury Land and Water Regional Plan, Environment Canterbury Regional Council uses the RMA river definition, and defines drain as "*any artificial watercourse that has been constructed for the purpose of land drainage of surface or subsurface water and can be a farm* *drainage channel, an open race or subsurface pipe, tile or mole drain, or culvert*" (Environment Canterbury Regional Council 2018). Rules that relate to activities in drains are much more permissive than rules that relate to streams: thus, there is a legal advantage of classifying a drain in this way. However, these drains often form the headwaters of larger river systems that are highly valued. One example of this, that has received much media attention of late, is the Waikirikiri Selwyn River, which flows through Coes Ford and is a major tributary to Te Waihora Lake Ellesmere. Without addressing activities occurring in and around the smaller feeder drains, activities to clean up Coes Ford and Te Waihora Lake Ellesmere are set to fail. The reclassification of drains as streams would make these impacts unacceptable in the eyes of landowners, management agencies and the public.

The use of these techniques to remove excessive macrophyte growth and maintain drainage function each growing season does not seem an effective solution. Macrophyte biomass is removed; however, we are not effectively solving the problem as macrophytes quickly re-establish and management needs to be undertaken each growing season (or sometimes more frequently). In fact, managers are reinforcing the macrophyte growth problem by continuing to manage streams in this way, as repeated clearance selects for those species that are able to rapidly regrow, and over time communities become less diverse (Greer et al. 2012). Macrophyte clearance continues to occur because that is what has always been done (G. Bennett, personal communication). Attitudes to macrophyte control need to change for long-term solutions to be developed.

> *"If you always do what you've always done, you'll always get what you've always got"*
>
> \- Henry Ford

Investigating alternative macrophyte control tools

Given the limitations of these conventional techniques, investigations into alternative control tools are warranted. I have undertaken small- (< 5 m), reach- (50 m) and large-scale trials (up to 400 m), evaluating the effectiveness of various conventional and alternative macrophyte control tools, including: hand weeding, herbicide spray, weed mat, flower and seed removal, shading, physical disturbance and sediment removal (Chapters 3 and 4).

My small-scale trial was beneficial in terms of informing which control tools were effective and warranted further testing at the reach- to large-scale. The 2 m x 2 m plots were an appropriate scale for undertaking treatments and measuring effectiveness; however, they were challenging in terms of edge effects and macrophyte invasion from adjacent areas. This trial was also only undertaken in a single stream which had been “restored” through bank re-battering and riparian planting. As a result, macrophyte growth along the length of the experimental reach was uneven. However, the blocked, randomised replication of treatment attempted to account for this. This trial identified hand weeding, weed mat and full shading as effective control techniques worth scaling up to a reach- to large-scale (Chapter 3).

While effective both in the short-term and at a small-scale, I did not find large-scale intensive hand weeding to be an effective long-term macrophyte control option. However, hand weeding has been used to successfully remove unwanted and undesirable species while leaving the rest of the community intact (Chisholm 2006; Bellaud 2014). This technique is very labour intensive and expensive, and ongoing surveillance and maintenance weeding is required (Hussner et al. 2017).

Unsurprisingly as a commonly used management option, glyphosate spray was found to be an effective macrophyte control technique, however macrophytes begin to recover within weeks to months. The use of chemical herbicides has been attracting increased public interest and concerns have been raised about the toxic effect of glyphosate on aquatic life (Brooker and Edwards 1975; Kelly et al. 2010; James 2011). There are also concerns of secondary effects including depleted dissolved oxygen levels and release of nutrients from decomposing plants, and sudden changes in habitat influencing refugia and food sources for aquatic invertebrates and fish fauna (Jewell 1971; Brooker and Edwards 1975; James 2011) although scientific evidence is limited. Public concern in Christchurch has resulted in the Christchurch City Council committing to limiting the use of glyphosate-based sprays to areas with no public access or where there are no other alternatives. Since this decision, pressure has been placed on the Waimakariri District Council to reduce their use of glyphosate sprays.

To respond to these public concerns, alongside my thesis I undertook an investigation for the Waimakariri District Council (WDC) to understand the persistence of glyphosate in stream water and sediment and its short-term effects on freshwater invertebrates and fish following the spraying of waterways (unpublished report to WDC; included as supplement to Chapter 5).

I found that glyphosate and AMPA (the primary breakdown product of glyphosate) were already present in the sediment in both control and spray reaches before spraying even started (Collins and Harding 2017). This implies that parties other than the Council are spraying waterways or nearby areas, and this makes determining the effects of spraying on animal life in these waterways difficult. Glyphosate and AMPA were present in the water column for 1-2 days following spraying, but glyphosate quickly bound to sediment and broke down to AMPA. Glyphosate and AMPA were still present in the sediment at both the control and spray reaches 14 weeks after spraying. Macrophytes in the spray reaches were greatly reduced by glyphosate, being reduced from 90% cover to 20%, however 14 weeks after spraying macrophyte cover in these reaches had returned to about 50%. I could not detect any effect of glyphosate on stream invertebrate species richness, metrics such as the MCI and SQMCI or fish (Collins and Harding 2017). However, these waterways are highly modified environments, and invertebrates and fish that occupy them are tolerant of water quality in these systems. This study suggested that the use of glyphosate spray to control macrophytes has minimal effect on the aquatic environment, however macrophytes were growing back within three months of spraying.

Rethinking riparian buffer design

Shading with ca. 80 % light reduction reduced both macrophyte height and cover (Chapter 2) and polythene shading was effective for controlling submerged bed macrophytes in Harris Drain (Chapter 4). Unsurprisingly, I am not advocating for the construction of shade tunnels over or floating of polythene down hundreds of kilometres of lowland streams, but an obvious way to achieve this shading is through riparian planting. While restoration of riparian planting has been promoted as a means of shading to regulate stream temperatures (Kauffman and Krueger 1984; Collier et al. 1995), it is not often thought of as a means of nuisance macrophyte control. I suggest that rethinking riparian buffer design could result in additional benefits to aquatic systems by controlling nuisance aquatic macrophytes (Chapter 4). Riparian planting is more effective at providing shading in small to medium sized channels (Davies-Colley et al. 2009; O'Briain et al. 2017; Willis et al. 2017), and stream orientation is also important due to the angle and the path of the sun relative to the channel (Rutherford et al. 1997).

I found weed mat to be a novel and effective means of controlling sprawling emergent macrophyte species. In my trials, weed mat was placed 1 m up the stream bank extending down the bank into the waterway for a further metre. This appears to be the zone that monkey musk and watercress seedlings first establish, with their growth then extending out across the channel. I have found that some types of weed mat performed better in the aquatic environment than others (Chapter 3, Collins et al. 2018a). The biodegradable wool weed mat had a very limited lifespan on stream banks (<3 months). Coir coconut fibre matting allowed the establishment of monkey musk and watercress through its coarse weave. In contrast, plastic woven weed mat was a more robust hard-wearing and long-lived product. A biodegradable material would be the ideal alternative in this situation as it would not require removal once plantings have established. This leaves the potential for the development of a novel material that has a tight weave and slow breakdown rate.

The combination of both weed mat and shading potentially provides an even more effective macrophyte control method. Using weed mat when establishing riparian planting occurs is not a novel idea. When planting is undertaken, often seedlings are protected by CombiGuards, a 300 mm high plastic sleeve supported by four bamboo stakes, with a 200 mm square of weed mat covering the soil. CombiGuards protect the newly planted seedlings from weed competition, animal browsing, maintenance spraying and extreme weather. However, covering stream banks in weed mat and cutting holes to plant seedlings into is an unconventional technique I am proposing for aquatic weed control. Weed mat is a practical and effective short-term solution while riparian plantings establish; then, in the longer term, riparian plants will establish and grow up, providing shade to support macrophyte control. Weed mat has the additional benefits of stabilising bare soil, retaining soil moisture and preventing nuisance terrestrial weeds from establishing in newly planted riparian buffers. Consideration needs to be given to placement of riparian plant species that will grow up to provide shading, as these should be sited to provide maximum shade across the stream channel. Several riparian planting guides for Canterbury exist, offering guidance and recommending suitable species to plant at different zones within the riparian cross section (Christchurch City Council 2005; Environment Canterbury Regional Council 2011; DairyNZ 2014). Bank rebattering prior to planting may be required where stream banks have become over-steepened and banks are eroding. Rebattering involves earthworks to reduce bank slope and stabilise the bank. It is prudent to rebatter (when required) prior to planting, to address sediment sources prior to investment of significant funds and energy. Another often overlooked factor, is the placement of the bottom row of plants in relation to the edge of the stream. The bottom row of *Carex* plants should be placed as close to the edge of the stream wetted cross section as is practical, to ensure maximum benefit of shade provided by overhanging riparian vegetation is reached in the shortest possible time. Undertaking riparian restoration in this way will aid the recovery of instream habitat, by providing habitat, organic matter inputs and shading in addition to the riparian buffer, helping to restore the balance in stream systems and reduce the need for costly management intervention.

In addition to the effectiveness of alternative macrophyte control techniques, the cost of the various techniques and the frequency they are required are also important. The alternative macrophyte control techniques I have shown to be effective in my thesis have higher immediate costs (Table 5.1). However, the frequency that traditional techniques are required to maintain control suggests that the cost can balance out over time. Additionally, going further than the monetary cost of techniques and considering the costs and benefits in ecological terms further promotes the use of alternative control methods.

Table 5.1. Approximate costs of macrophyte control techniques.

	Control technique	Approximate cost per metre	Frequency required
Conventional	Mechanical	$0.36*	1-3 x per year
	Herbicide	$0.25*	1-3 x per year
	Hand weeding – cutting with sickle or scythe	$3.75*	1-3 x per year
Alternative	Weed mat	$4	One-off
	Hand weeding – removal of all plant biomass	$5	Ongoing
	Planting	$7	One-off plus maintenance
	Planting and weed mat	$10	One-off

* indicates cost taken from Hinds Drains Working Party (2016).

Opportunities for future development

I have highlighted the effectiveness of shading at relatively high levels to control sprawling emergent macrophytes in small agricultural streams, however more research is required around the optimal levels of shading required to control different macrophyte species and the levels of shading that can be provided by riparian planting with different species makeup. In addition, I have also demonstrated the effectiveness of weed mat on stream banks to control sprawling emergent macrophyte growth, but the lack of a durable weed mat material that biodegrades means that the plastic alternative needs to be removed. There is potential for the development of an innovative weevvvvd mat material that has a tight weave and slow breakdown rate, eliminating the need for weed mat removal post macrophyte control. Finally, further research on the ability of artificial disturbance by creating high flow events to manage macrophyte growth could be explored. There is high potential to undertake this in conjunction with irrigation schemes operating around New Zealand.

In conclusion, my research has contributed to understanding factors that affect macrophyte diversity and abundance in lowland agricultural streams and considered alternative macrophyte control options. My results suggest that alternative and less environmentally-damaging macrophyte control options exist, and while costs are greater in the short-term, their long-term effectiveness mean that over time, costs balance out. I hope that these findings will be adopted for the management of macrophytes so that, in time, the need for ongoing maintenance by mechanical clearance or herbicide spray is reduced.

Supplement to Chapter 5:
Persistence and ecological consequences of glyphosate to control aquatic weeds in Waimakariri lowland waterways. Unpublished report prepared by Katie Collins and Jon Harding for the Waimakariri District Council, presented to the Council Meeting on 24 October 2017

Canterbury Waterway Rehabilitation Experiment

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UNIVERSITY OF CANTERBURY
Te Whare Wānanga o Waitaha
CHRISTCHURCH NEW ZEALAND

Persistence and ecological consequences of glyphosate to control aquatic weeds in Waimakariri lowland waterways

Katie Collins, Jon S. Harding
Corresponding author: carex@canterbury.ac.nz

September 2017

Executive Summary

This study and report was undertaken by researchers from CAREX and no payment was received for this work. Waimakariri District Council paid for commercial analysis of glyphosate and AMPA. The purpose of this study was to understand the persistence of glyphosate in stream water and sediment and its short-term effects on freshwater invertebrates and fish following spraying of waterways.

From December 2016 – March 2017 five waterways near Rangiora were investigated to test the effect of glyphosate on aquatic weeds, stream invertebrates and fish. In each waterway an upstream reach was left as an unsprayed control and a downstream reach was sprayed. Samples were collected in each reach before and after spraying. Glyphosate and AMPA (the product of glyphosate) were already present in the sediment at both the control and spray reaches before spraying even started. This implies that parties other the Council are spraying waterways or nearby areas, and this makes determining the effects of spraying on animal life in these waterways difficult.

Glyphosate and AMPA were present in the water column for 1-2 days following spraying, but glyphosate quickly bound to sediment and broke down to AMPA. Glyphosate and AMPA were still present in the sediment at both the control and spray

reaches 14 weeks after spraying. Weeds in the spray reaches were greatly reduced by glyphosate, being reduced from 90% cover to 20%, however 14 weeks after spraying weed cover in these reaches had returned to about 50%. We could not detect any effect of glyphosate on stream invertebrate species richness, metrics such as the MCI and SQMCI or fish. These waterways are highly modified environments, and invertebrates and fish that occupy them are tolerant of water quality in these systems. Given the small sample size (five waterways), the findings of the study are limited and add to our understanding of drain maintenance on aquatic systems.

1. Introduction

Excessive growth of aquatic macrophytes (weeds) is a significant problem in lowland agricultural waterways, including in the Waimakariri District. Management is undertaken by Councils to ensure drainage is maintained, most commonly using mechanical clearance, herbicide spray and hand weeding.

Glyphosate is one of the world's most effective and most frequently used herbicides. It is a non-selective, broad-spectrum herbicide commonly used on emergent (surface dwelling) and marginal (bankside) macrophytes, but following manufacturers instructions, spraying directly on the waterway should be minimised.

Concerns have been raised about the toxic effect of glyphosate on aquatic life. There are also concerns of secondary effects including depleted dissolved oxygen levels and release of nutrients from decomposing plants, and sudden changes in habitat influencing refugia and food sources for aquatic invertebrates and fish.

To respond to public concerns, an investigation was carried out by the University of Canterbury on behalf of the Waimakariri District Council on the use of glyphosate spray to control aquatic macrophytes. This investigation was undertaken between December 2016 and March 2017.

The aims of this study were to investigate:

- the persistence of glyphosate in the stream water and sediment following spraying
- the effect of glyphosate on the freshwater invertebrates and fish in sprayed waterways

2. Methods

2.1. Experimental design

The impact of glyphosate was tested in five waterways. In each waterway an upstream 200m reach was selected which was not sprayed (control reach) and a 200m reach downstream was sprayed (treatment reach). The five waterways were scheduled to be sprayed by the Waimakariri District Council as part of their annual weed control program. They were:

- Ashworths: Ashworths Road Drain, between Mill Road & Main Drain Road
- Ohoka: Ohoka Stream North Branch, between Mill Road & the first gate along the walkway
- Threlkelds: Threlkelds Road, upstream of Main Drain Road
- Easterbrook: Easterbrook Road, upstream of Hicklands Road
- Ashby's: No. 4 Drain, upstream of Hicklands Road

Figure 1: Location of the five waterways used in the spray trial.

A 200m stretch at the top of each reach was left unsprayed as a control reach. Macrophytes were sprayed from the 200m point downstream. Sampling of the control reach was undertaken 100m into the reach, and the spray reach was sampled at 400m (Fig 2).

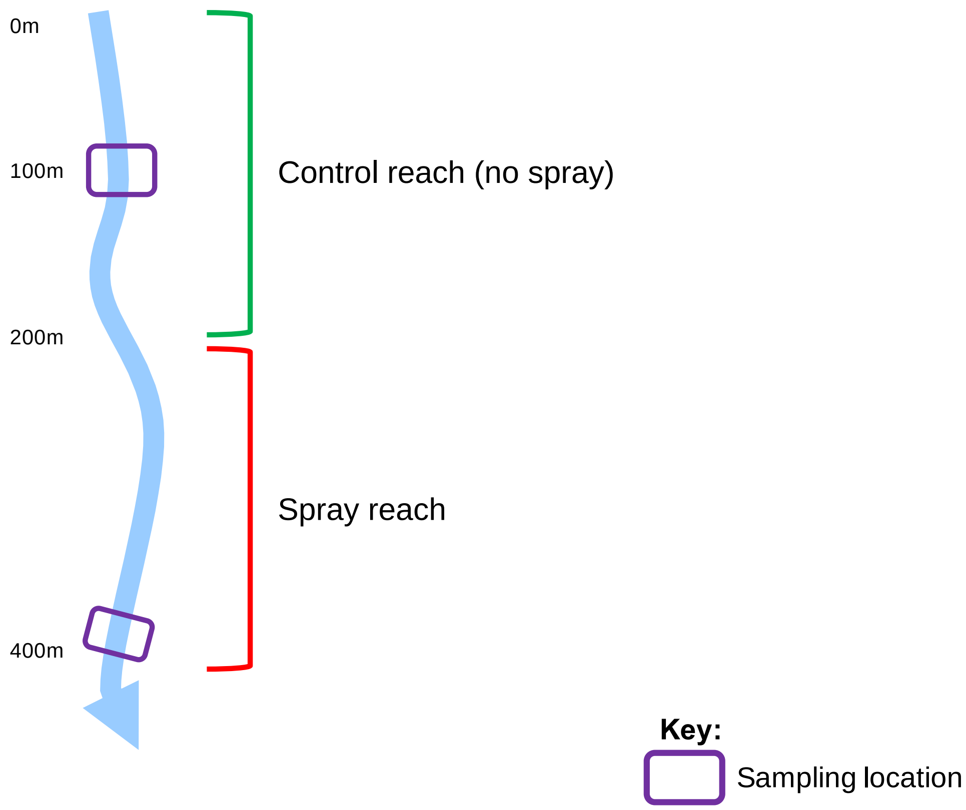

Figure 2: Spray trial experimental design used in all five waterways

Spraying was carried out by the Waimakariri District Council's contractor on 21 December 2016.

2.2. Weed monitoring

At each of the control (100m) and spray (400m) reaches, three macrophyte assessment cross-sections were set up. These cross-sections were measured before the spray trial (pre-spray), and 3, 6 and 14 weeks after spraying (post spray).
On each cross-section, aquatic weed species and the height above the water surface were recorded every 10cm across the wetted width of waterway.

2.3. Glyphosate and AMPA sampling of water and sediment

When glyphosate contacts water, there are two major pathways of dissipation: binding to sediments, and microbiological breakdown. When sediments are present glyphosate rapidly binds to soil particles, bacteria and fungi in the water and sediment also breakdown glyphosate into aminomethylphosphonic acid (AMPA). AMPA can remain stable in sediments for some time. We measured both glyphosate and AMPA to better understand the persistence and breakdown time in these streams and sediments.

Glyphosate and AMPA samples of both stream water and stream bed sediment were collected and sent for analysis by AsureQuality (Wellington).

Water samples were collected pre-spray, the day of spraying (both control and spray reaches) and 1 and 5 days post spray (spray reaches only).

Samples of sediment were collected pre-spray (control and spray reaches) and 5 days, 3 weeks (spray reaches only) and 6 weeks post spray (control and spray reaches).

2.4. Aquatic invertebrates

Aquatic invertebrates were collected at both control and spray reaches pre spray, 5 days and 6 weeks after spraying. In each reach a single invertebrate kick-net sample (500 µm mesh) was collected from five representative micro-habitats within the reach using the standard New Zealand protocols (Stark et al 2001). Samples were labelled and stored in 70 % ethanol.

In the laboratory the samples were sieved (500 µm Endecott sieve), and all invertebrates identified to the lowest practicable level (usually genus) using identification guides (such as Winterbourn 2006). Coded abundances of taxa were recorded as described by Stark (1998).

We then calculated several stream health metrics to determine the impact of the spray trial on aquatic invertebrates. The Macroinvertebrate Community Index (MCI) uses the presence or absence of taxa and their tolerance to pollution to indicate stream health. The MCI ranges from 0 – 200, scores of less than 80 indicate a severely polluted system while scores over 120 are considered healthy (Table 1). A second metric called the Semi-Quantitative Macroinvertebrate Community Index (SQMCI) was calculated using the pollution tolerances of taxa present and the coded abundance data. SQMCI's range from 0 – 10. Values less than 4 indicate a severely polluted system while values more than 6 indicate health systems.

Table 1: Interpretation of MCI and SQMCI values.

Water quality	Description	MCI	SQMCI
Excellent	Clean water	> 119	> 5.99
Good	Doubtful quality or possible mild pollution	100 – 119	5.00 – 5.90
Fair	Probable moderate pollution	80 – 99	4.00 – 4.99
Poor	Probable severe pollution	< 80	< 4.00

2.5. Fish sampling

Freshwater fish were sampled with a portable (KAINGA EFM300) electric fishing machine by spot fishing in areas where aquatic weed cover was less than 40%. Electric fishing was undertaken at both control and spray reaches pre spraying and 3, 6 and 14 weeks post spray. However, this was problematic especially prior to spraying as weed cover was extensive and the high weed cover potentially confounded any results. Captured fish were identified to species level where possible in the field. Very small fry (< 4 cm) were identified to family. Glass eels and elvers (Anguillidae) (< 10 cm) were recorded as elvers.

Table 2: Timing of different sample collection over the experimental period.

Days since spraying	**Water samples**		**Sediment samples**		**Macrophyte transects**	**Aquatic Invertebrates**	**Fish**
	Control	**Spray**	**Control**	**Spray**	**Control & Spray**	**Control & Spray**	**Control & Spray**
Pre spray							
Day of spray							
Spray 1 day							
Spray 5 days							
Spray 3 weeks							
Spray 6 weeks							
Spray 14 weeks							

3. Results

3.1. Glyphosate and AMPA in water

Prior to spraying no glyphosate was detected in the water but AMPA was found in water in the control sites. No glyphosate or AMPA were present in the water on the day of spraying at any control (non-sprayed) reaches (Fig 3A & B) whereas both glyphosate and AMPA were present in the water on the day of spraying at all spray (treated) reaches. On the day after spraying, glyphosate was detected in the water at all spray reaches at low concentrations. AMPA was only detectable in the water at the Easterbrook spray reach (Fig 3A & B). Five days after spraying, glyphosate and AMPA were both virtually undetectable in the water at all spray reaches (Fig 3A & B).

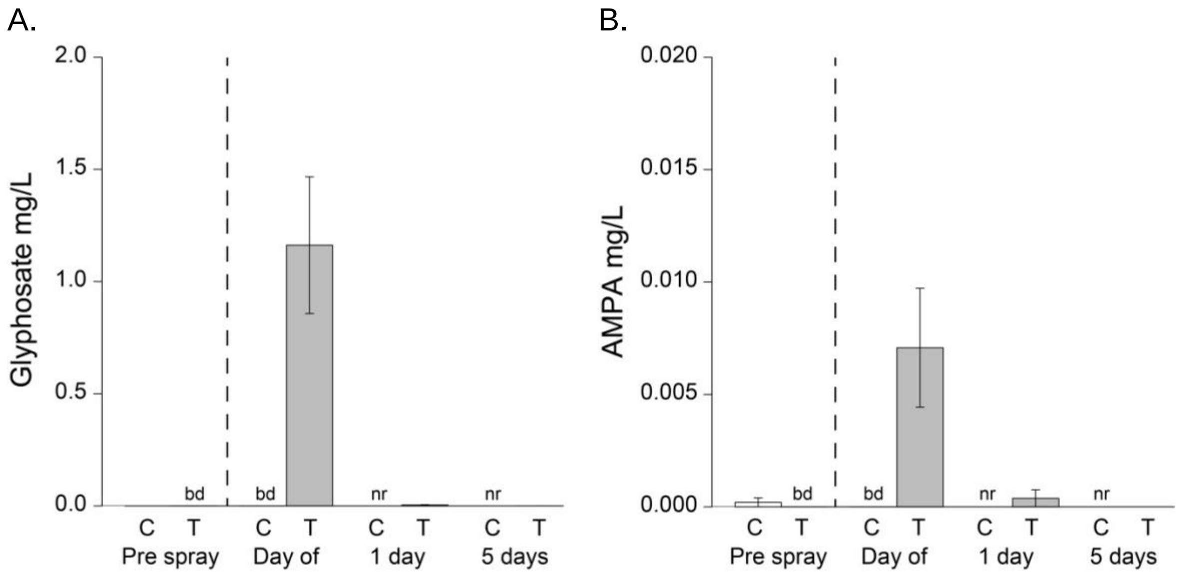

Figure 3: A. Mean glyphosate and B. Mean AMPA concentrations in water pre spraying, on the day of spraying, the day after spraying and 5 days after spraying. Control reaches are shown in white, treated (spray) treated reaches are shown in grey. Time of spraying is indicated by the dotted line. nr = sample not run, bd = sample result below detectable limit. Mean values are shown with ± 1 Standard error.

3.2. Glyphosate and AMPA in sediment

Pre spraying, glyphosate and AMPA were detected in the sediment in both control and spray reaches (Fig 4A & B). Six weeks after spraying, glyphosate and AMPA were still detectable in the sediment in both control and spray reaches (Fig 4A & B).

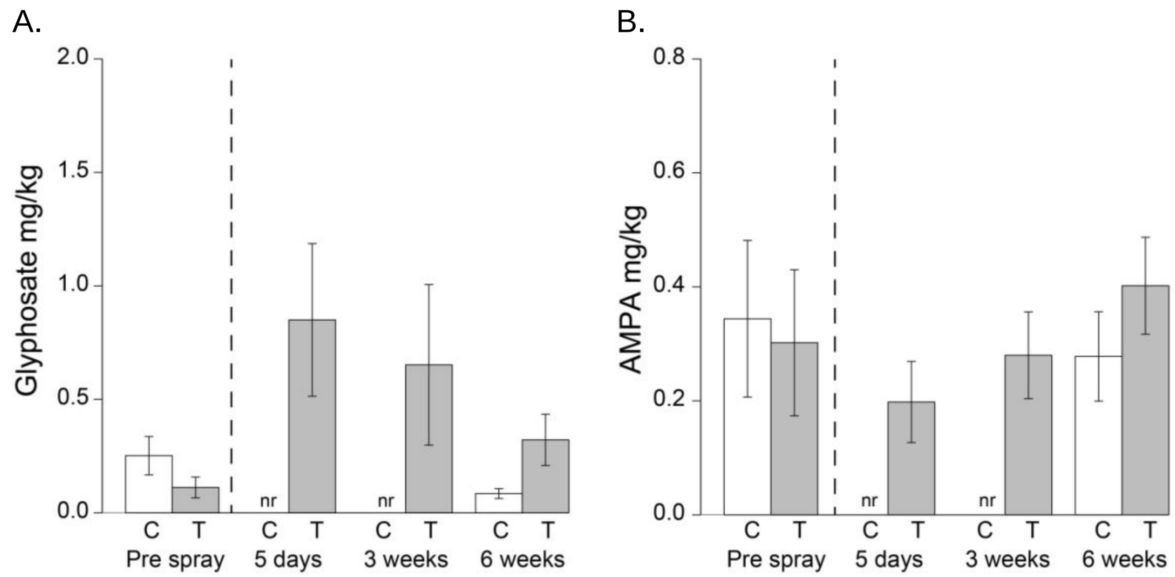

Figure 4: A. Mean glyphosate and B. Mean AMPA concentrations in sediment pre spraying, 5 days after spraying, 3 weeks and 6 weeks after spraying. Control reaches are shown in white, treated (spray) treated reaches are shown in grey. Time of spraying is indicated by the dotted line. nr = sample not run, bd = sample result below detectable limit. Mean values are shown with ± 1 Standard error.

3.3. Aquatic weed cover

Macrophyte cover was between 80 – 100 % pre spraying. Three weeks post spraying, macrophyte cover was greatly reduced in the spray reaches (Fig 5, Photos 1-3). Fourteen weeks post spraying, macrophytes were starting to grow back in sprayed reaches (Fig 5).

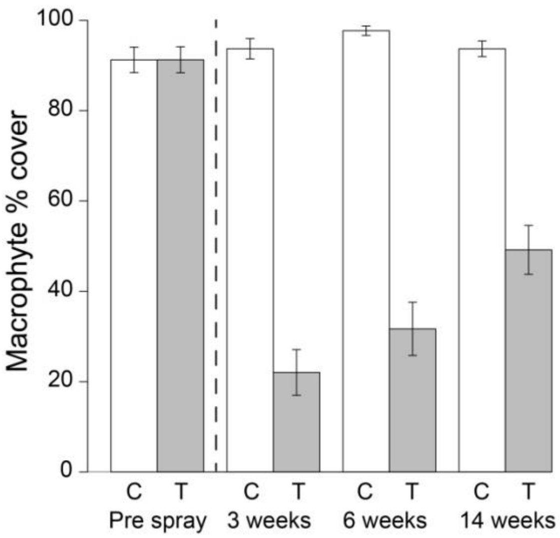

Figure 5: Mean macrophyte percent cover pre spraying, 3 weeks, 4 weeks and 14 weeks after spraying. Control reaches are shown in white, treated (spray) treated reaches are shown in grey. Time of spraying is indicated by the dotted line. Mean values are shown with ± 1 Standard error.

Photo 1: Threlkelds Road site pre spraying

Photo 2: Threlkelds Road control site 3 weeks after spraying

Photo 3: Threlkelds Road spray site 3 weeks after spraying

3.4. Invertebrate species richness, MCI and SQMCI

We compared mean values for invertebrate species richness, MCI and SQMCI and found no difference, suggesting these communities are not affected by the presence of glyphosate in the water or sediment (Fig 6A, B & C). MCI and SQMCI scores at all sites indicated probable moderate levels of pollution.

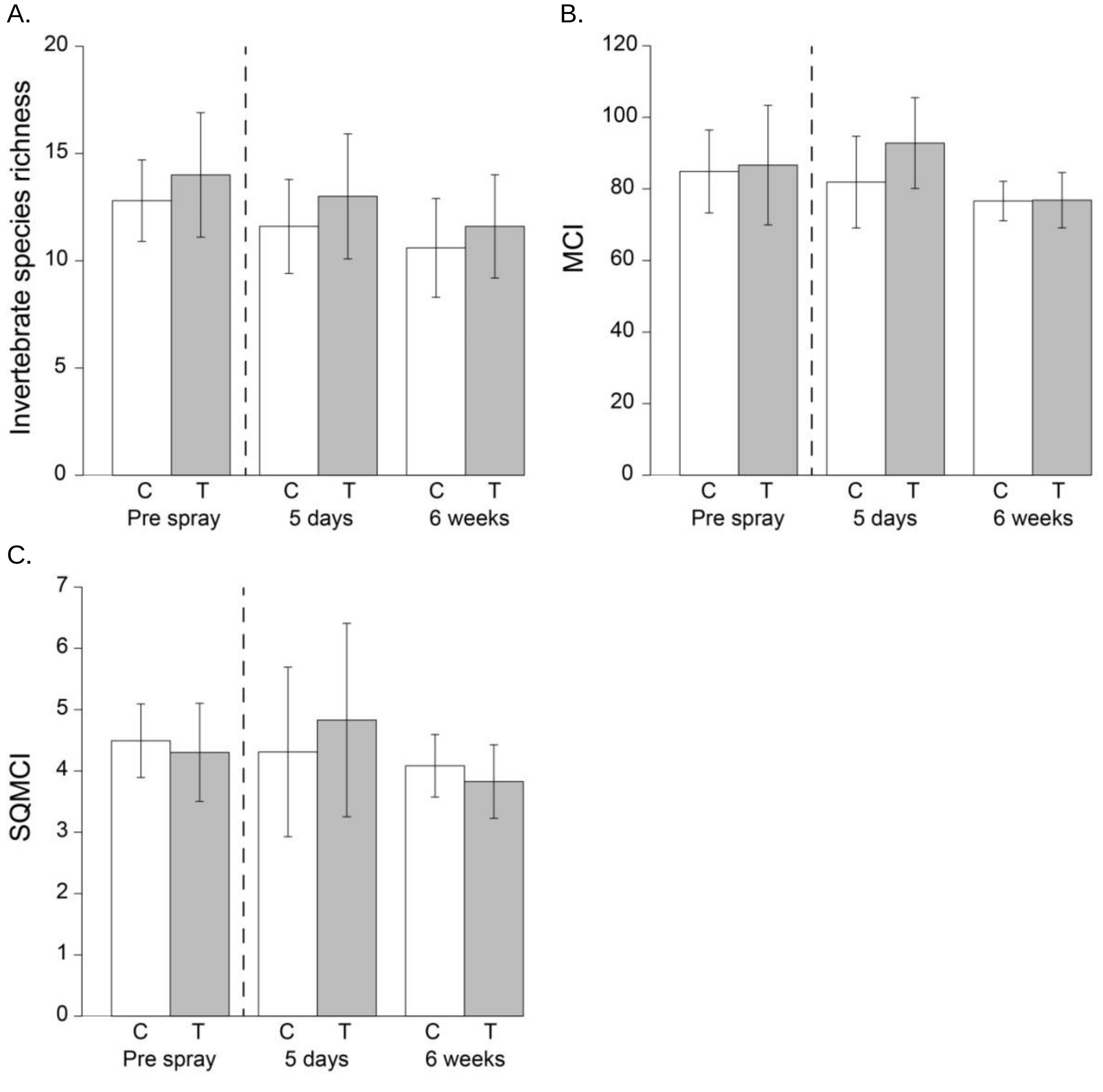

Figure 6: A. Mean invertebrate species richness, B. MCI and C. SQMCI pre spraying, 5 days and 6 weeks after spraying. Control reaches are shown in white, treated (spray) treated reaches are shown in grey. Time of spraying is indicated by the dotted line. Mean values are shown with ± 1 Standard error.

3.5. Fish species richness

Five fish species were observed in the five waterways, including: upland bullies (*Gobimorphus breviceps*), common bullies (*Gobimorphus cotidianus*), shortfin eels

(*Anguilla australis*), one longfin eel (*Anguilla dieffenbachii*) and juvenile brown trout (*Salmo trutta*).

Post spraying no differences were observed in fish species richness despite a declining trend. It seems unlikely individual fish species were directly impacted (Fig 7). Unfortunately, the high weed cover made accurate fish data difficult to collect.

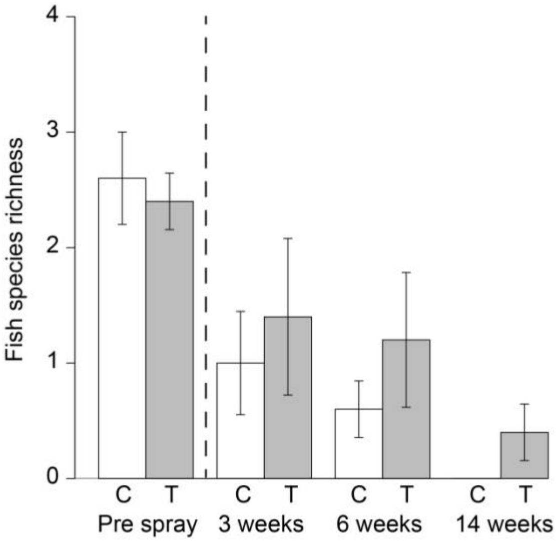

Figure 7: Mean fish species richness pre spraying, 3 weeks, 6 weeks and 14 weeks post spraying. Control reaches are shown in white, treated (spray) treated reaches are shown in grey. Time of spraying is indicated by the dotted line. Mean values are shown with ± 1 Standard error.

4. Final comments

- The purpose of this study was to understand the persistence of glyphosate in stream water and sediment and its short-term effects on freshwater invertebrates and fish following spraying of waterways.
- Glyphosate and AMPA were present in the water column for 1-2 days following spraying, but glyphosate quickly bound to sediment and broke down to AMPA
- Glyphosate and AMPA were already present in the sediment at both the control and spray reaches before spraying even started.
- Glyphosate and AMPA were still present in the sediment at both the control and spray reaches 14 weeks after spraying
- Spraying with glyphosate is an effective way to control aquatic weeds, however effectiveness is short lived and grow back is evident within three months
- Species richness of invertebrates and fish, MCI and SQMCI are not affected by the use of glyphosate to control emergent macrophytes. These drains are highly modified environments, and invertebrates and fish that continue to occupy them are tolerant of water quality in these systems.
- Glyphosate is commonly used for domestic purposes on lawns and gardens, and in agricultural landscapes. There are several ways it can enter waterways, including spray drift and direct runoff from sprayed land.
- This study was not designed to detect the sources of glyphosate in these stream systems. Our results show that either: glyphosate can persist in these systems between periods of drain maintenance, or the glyphosate in the system prior to commencement of this study was from other nearby sources.

5. Acknowledgements

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Plate 6: Sampling macrophytes in a Canterbury lowland agricultural stream.

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"Rivers know this: there is no hurry.
We shall get there some day."
– A.A. Milne


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